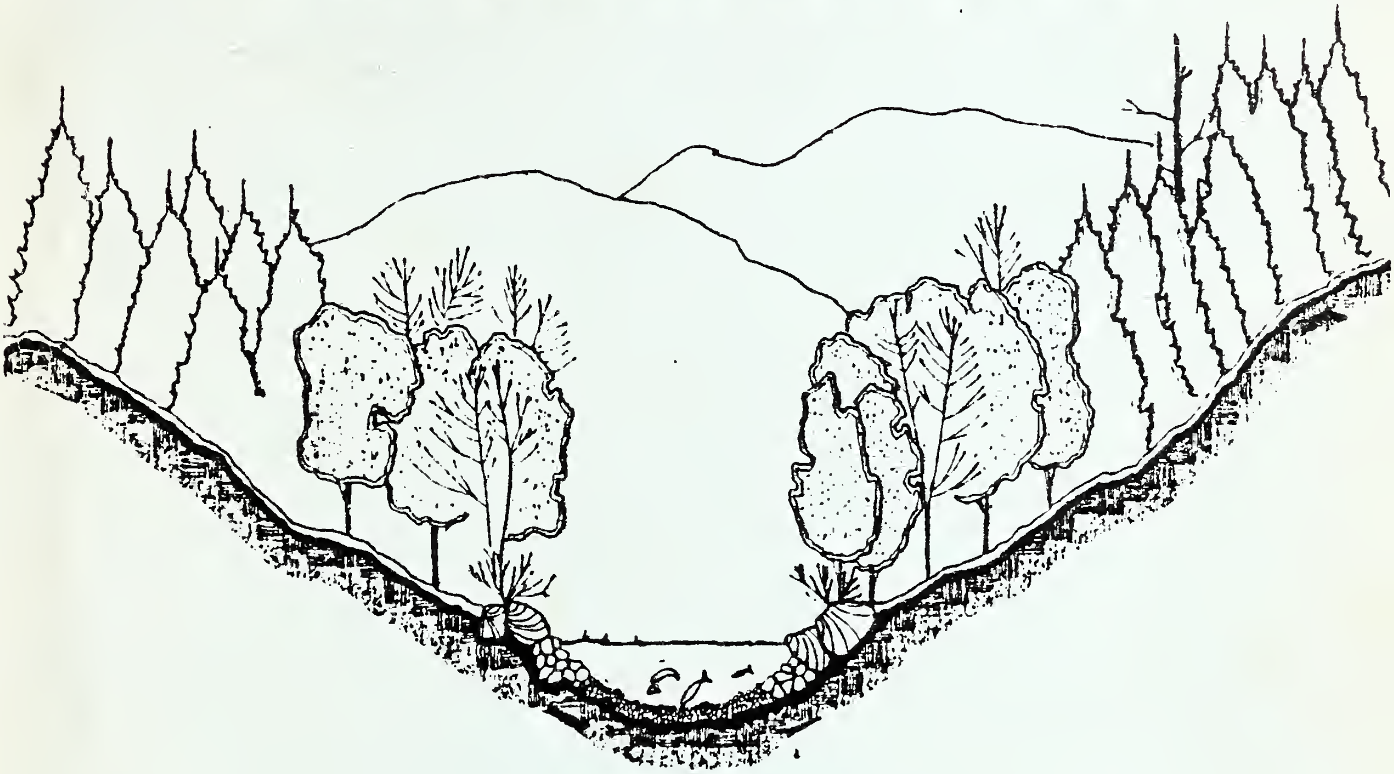
WATERSHED AND FISHERIES RESEARCH AT PNW



AN UPDATE

WITH EMPHASIS ON SOUTHWESTERN OREGON AND NORTHERN CALIFORNIA

Forest Service USDA

24500

WATERSHED AND FISHERIES RESEARCH

AT THE PACIFIC NORTHWEST FOREST AND RANGE EXPERIMENT STATION

An update with emphasis
on Southwestern Oregon and
Northern California

Grants Pass, Oregon

February 23 - 24, 1983

Co-sponsored by
Pacific Northwest Region
and
Pacific Northwest Forest and Range Experiment Station
USDA-Forest Service

Organized by

Mark Anderson, Siskiyou National Forest, Grants Pass, Oregon
Logan Norris, Forestry Sciences Laboratory, Corvallis, Oregon

828433

AGENDA

WATERSHED MANAGEMENT IN THE COASTAL ECOSYSTEM

FEBRUARY 23 and 24, 1983

FEBRUARY 23	SUBJECT	SPEAKER
	Anderson - moderator	
0930	Welcome and NFS Perspective	Bill Covey
0945	PNW and Watershed Research Perspective	Logan Norris
1015	BREAK	
	RIPARIAN ECOSYSTEM SESSION	
1030	Ecosystem, Structure, Function, and Research	Jim Sedell and Fred Swanson
1200	LUNCH	
	FISHERIES SESSION	
1300	Regional Fisheries Program	Gordon Haugen
1330	Anadromous Fishery Habitat	Fred Everest
1430	BREAK AND DISCUSSION	
1500	Riparian Zone and Large Organic Debris Management	Jim Sedell and Fred Swanson
1615	Panel Discussion	
1700	END OF DAY	
1830	Social Hour, Dinner, and Speaker	Dr. Larry Helms

AGENDA

FEBRUARY 24	SUBJECT	SPEAKER
	SOIL AND WATER SESSION	
0745	Meurisse Regional ~~Soil and Water~~ Perspective	Bob Meurisse and Gerald Swank , Smiv.
0815	Mass Soil Movement	Fred Swanson
0915	Maintenance of Site Productivity	Dick Miller
1000	BREAK	
1030	Effects of Logging and Associated Activities of Hydrologic Processes	Dennis Harr
1130	Panel Discussion	
1200	LUNCH	
1300	Swank Herbicides in Soil and Water	Logan Norris
1415	BREAK	
1445	On-going PNW-NSF Technology Transfer and Critique of Session	Logan Norris John Berry ~~Fred Everest~~ R.O. - Meurisse Swank
1345	END OF SESSION	

Thanks to Siskiyou

PNW and Watershed Perspectives

Logan A. Norris

I. Research and who does it in forestry

II. PNW in the Forest Service

III. PNW areas of responsibility

   A. Geographic areas

   B. Subject matter areas

   C. Line item integrity

IV. PNW structure

   A. Administrative

   B. Research locations

   C. Research work units

      1. Missions

      2. Project leaders and scientists

V. Formulation of watershed research programs

   A. Research work unit description

   B. Problem analysis

   C. Study plans

   D. Cooperative research

   E. Administrative studies

VI. Reporting watershed research results

   A. Publications

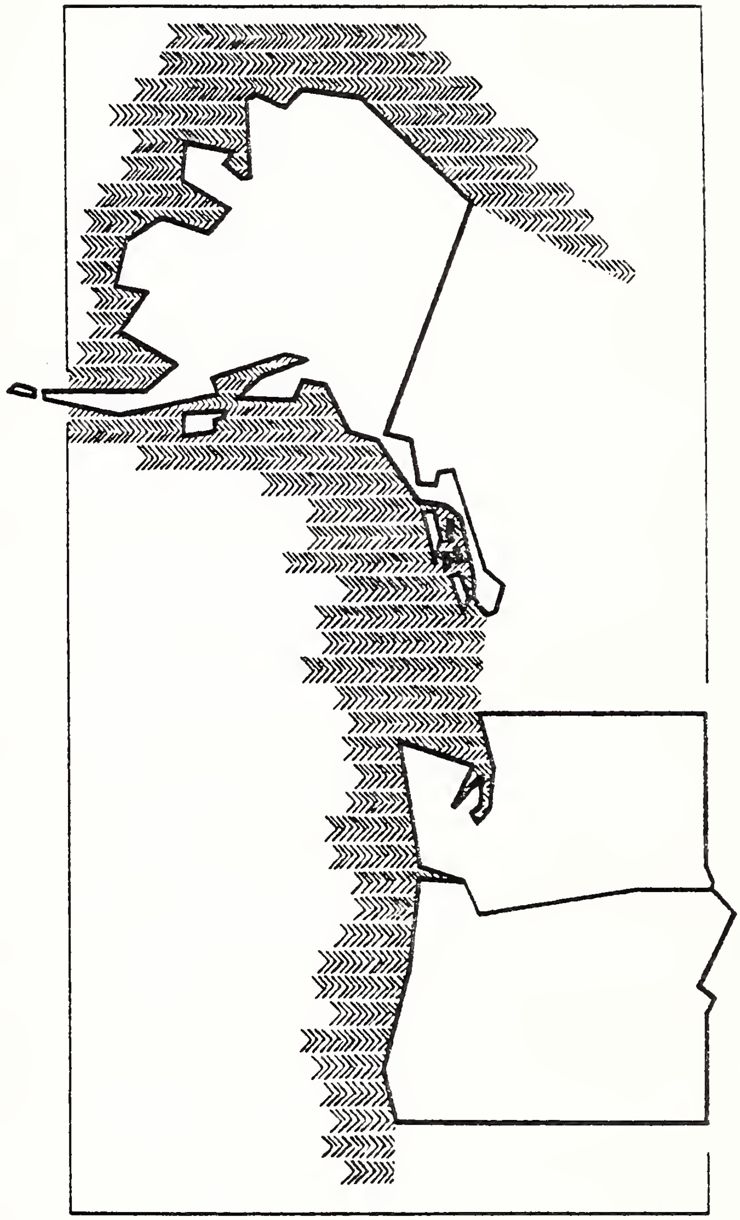
   B. Scientific and professonal meetings

C. Technology transfer and training sessions

D. Key contacts

E. Early innovators

Directory of Research Programs



United States Department of Agriculture

PREPARED BY Forest Service April 1982

Pacific Northwest Forest and Range Experiment Station

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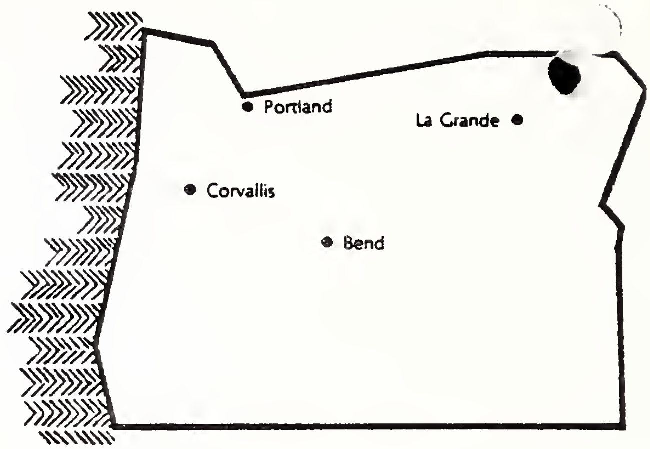
Requests for publications (ext. 2072)

General information (ext. 2094)

Current Investigations and Scientists in Charge



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Forest Residues and Energy Program, Research Unit 2107
Richard O. Woodfin Jr., Program Manager (ext. 2030)
- Wood for energy
- Residue measurement, management, utilization, and economic tradeoffs
- Prescribed fire as a management tool, emission sources and environmental effects (Seattle)
- Air resource management and planning
- Interior Alaska marketing, utilization, and woo[illegible] energy factors (Fairbanks)

Canada/United States Spruce Budworms Program—West, Research Unit 2211 (Operational responsibility for the western component of a cooperative program of the U.S. Department of Agriculture and the Department of Environment, Canada, is assigned to the Station.)
Ronald W. Stark, Program Manager (ext. 2034)
- Develop and improve methods of assessing and predicting budworm population levels and forest responses to budworm attacks
- Improve existing control technology and develop environmentally acceptable chemical, biological, and silvicultural treatment techniques

- Develop and improve methods for assessing and predicting short- and long-term impacts of budworms on forest resource uses and values
- Develop a pest management system for the western spruce budworm that can be integrated with current resource management systems

Timber Quality and Product Yield Potential of Western Softwood, Research Unit 3101
Thomas A. Snellgrove, Project Leader (ext. 2106)
- Product potential of standing dead western softwood timber
- Product yield research to reflect change in resource base, processing technology and utilization, and to develop analytical grading systems.

Renewable Resources Evaluation, Research Unit 4101
John H. Poppino, Project Leader (ext. 2117)
- Inventory renewable forest and related resources
- Evaluate current utilization of forest products
- Develop techniques for inventorying and evaluating forest and related resources
- Analyze future supplies of renewable resources
- Evaluate the effects of alternative levels of resource outputs on regional economies

Economics of Forest Land Management, Research Unit 4201
Roger D. Fight, Project Leader (ext. 2086)
- Economic foundations for major policy decisions involving forest land management
- Economic guides and scheduling techniques for timber management activities
- Economic relationships between production of timber and non-timber goods and services from forest land

Foreign Trade Analysis, Research Unit 4202
David R. Darr, Project Leader (ext. 2088)
- Economic foundations for major policy decisions involving international trade
- Consequences of foreign markets for wood products in the Pacific Northwest
- Economic information on western forest products market

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Don H. Boelter, Assistant Station Director (ext. 4381)

Reforestation Systems in the Pacific Northwest, Research Unit 1201
Peyton W. Owston, Project Leader (ext. 4343)
- Guidelines for establishment and crop tree release in plantations of the Coast Range
- Development of vegetation treatments and types of planting stock for reforesting harsh sites in sout western Oregon
- Determination of the nature and intensity of vegetativ competition

Ecological Basis for Management of Northwest Conifero Forests, Research Unit 1251
Jerry F. Franklin, Project Leader (ext. 4362)
- Management of upper-slope true fir-hemlock forests; regeneration, species selection, growth and yield
- Identification and evaluation of potential marginal commercial forest land (hot, dry and cold, snowy habitats)
- Management of wildland landscapes (National Park, wilderness, roadside)

Genetic Improvement of Douglas-fir and other Northwest Trees, Research Unit 1401
Roy R. Silen, Project Leader (ext. 4339)
- Seed orchard technology
- Advanced generation breeding questions
- Delimiting genetically safe movements of seed

Impacts of Management Activities on Watershed Resource Values in Douglas-fir Forests, Research Unit 1653

Logan A. Norris, Project Leader (ext. 4387)

- Effects of timber harvest on snowmelt during rainfall in the transient snowpack zone
- Soil and sediment movement in natural and managed forests and streams
- Effects of logging, residue treatment and revegetation on the amount, form, and distribution of nitrogen in managed forests
- Effects of brush control practices on nutrient relations and soil properties
- Offsite disposition and entry and fate of specific pesticides in air
- Long term outputs of water soil, and nutrients from undisturbed and logged watersheds

Behavioral and Microbial Agents for Managing Western Forest Insects, Research Unit 2204

Gary E. Daterman, Project Leader (ext. 4334)

- Identification of behavioral chemicals and microbial agents useful in integrated pest management
- Development of survey and monitoring systems which predict insect population changes and damage
- Development of behavioral chemicals and microbials as insect control agents

Forest Tree Diseases in the Pacific Northwest, Research Unit 2209

Earl E. Nelson, Project Leader (ext. 4416)

- Techniques for quantifying and predicting damage from laminated root rot
- Control of laminated root rot
- Etiology and effects of diseases of flowers, cones, and seed in established seed orchards
- Assessing growth loss caused by dwarf mistletoe

Mycorrhizal Applications in Ecosystem Manageme-**.** Research Unit 2210

James M. Trappe, Project Leader (ext. 4402)

- Mycorrhizal ecology and manipulation for improving forest regeneration success

Economics of Regional and Local Impacts of Forest Resource Management Decisions, Research Unit 4203

Con H. Schallau, Project Leader (ext. 4413)

- Developing cost-effective methods for estimating regional and local employment and income effects of forest resource management decisions
- Measuring local and regional effects of forest resources policy on public and private investment
- Examining the extent to which alternative public policies might affect economic stability of timber dependent communities

Bend, Oregon
Silviculture Laboratory
1027 NW Trenton Avenue, Bend 97701
Phone: 503/382-6922, ext. 283
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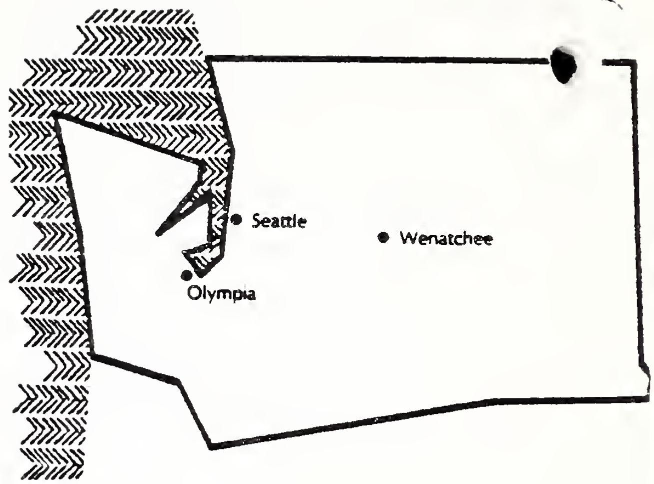
Culture of Forests of Eastern Oregon and Washington, Research Unit 1203
Robert E. Martin, Project Leader
- Yield and regeneration of ponderosa pine, lodgepole pine, and interior mixed-conifer forests
- Use and effects of fire for multiple use of east-side forests and ranges
- Fire-insect-disease interactions in east-side forests

La Grande, Oregon
Range and Wildlife Habitat Laboratory
C and Gekeler Lane
Rt. 2 Box 2315, La Grande 97850
Phone: 503/963-7122, FTS 423-4111

Wildlife Habitat and Range, Research Unit 1701
Jack W. Thomas, Project Leader
- Description of seasonal ranges for ungulates
- Management systems for ungulate ranges
- Management for non-consumptive uses of wildlife
- Alternative management strategies of forest-range grazing and consequent environmental effects, economic returns, and social benefits

Population Ecology and Integrated Pest Management Strategies for Western Forest Defoliators, Research Unit 2201
Boyd E. Wickman, Project Leader
- Causes and prevention of Douglas-fir tussock moth outbreaks
- Long-term effects of defoliation by Douglas-fir tussock moth on tree and stand growth
- Processes responsible for population changes of the western spruce budworm
- Introduced parasites for control of the larch casebearer in western North America



Olympia Washington
Forestry Sciences Laboratory
3625-93d Avenue, SW, Olympia 98502
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FTS 434-0470

Biology and Silviculture of Forests of the Douglas-fir Region, Research Unit 1207
Dean S. DeBell, Project Leader
- Control of tree-spacing and levels-of-growing-stock in young Douglas-fir
- Increasing site productivity through fertilization
- Management of biological nitrogen fixation in forest soils
- Silviculture of red alder, western redcedar, western hemlock, and true firs
- Potential yield of managed stands

Old-Growth Forest Wildlife Habitats, Research Unit 1706
Leonard F. Ruggiero, Program Manager
- Identification of animal and plant species dependent on or which find optimum habitat in old-growth forests
- Description, classification, and inventory of old-growth forest ecosystems
- Determine biological requirements and ecological relationships of species found in old-growth forests
- Evaluating old-growth management alternatives and their economic impacts

Seattle, Washington
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4507 University Way NE, Seattle 98105
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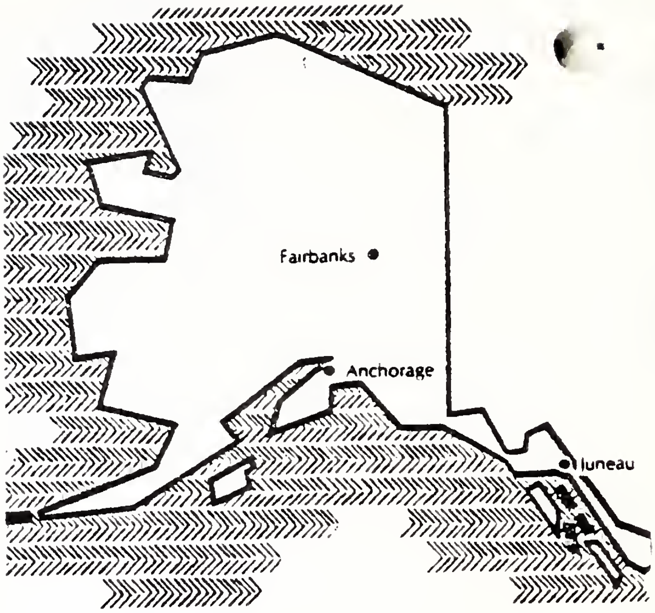
Public Uses of Wildlands, Research Unit 1901
Roger N. Clark, Project Leader (ext. 7817)
- Integrating recreational use of forest land with other multiple uses
- Reducing impacts of vandalism and other depreciative behavior in forest settings

Logging Systems for Fragile Mountain Terrain, Research Unit 3701
Charles N. Mann, Project Leader (ext. 7814)
- Total logging systems planning
- Advanced harvesting systems
- Residue handling and transportation systems
- Improved harvesting systems for Alaska

Wenatchee, Washington
Forestry Sciences Laboratory
1133 N. Western Avenue, Wenatchee 98801
Phone: 509/662-4315
FTS 390-0315

Productivity of High Elevation Forests of the Interior Northwest, Research Unit 1601
Paul J. Edgerton, Project Leader
- Maintaining and improving site productivity
- Developing strategies for revegetation/regeneration
- Assessing the effects of management activities on water quality



Fairbanks, Alaska
Institute of Northern Forestry
308 Tanana Drive, Fairbanks 99701
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FTS 907/456-0300, ask for 474-7433

Ecology and Management of the Taiga, Interior Al
Research Unit 1651
C. Theodore Dyrness, Project Leader

- Classification and characterization of major Alaskan vegetation units
- Behavior and effects of fire and development of improved fire management
- Management strategies for regenerating and growing shrubs and trees
- Interrelationships among bark beetle and defoliator populations and their hosts
- Hydrologic, sediment, and water quality relationships in managed vegetation types
- Characteristics and management of wildlife habitat

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Anchorage 99508
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Renewable Resources Evaluation for Alaska, Research Unit 4013
Vernon J. LaBau, Project Leader
- Coastal Alaska resources analysis
- Interior Alaska multi-resource assessment
- Economic analysis of timber resources

Juneau, Alaska
Forestry Sciences Laboratory
PO Box 909, Juneau 99802
Phone: 907/586-7301
FTS 907/586-7301

Ecology of Southeastern Alaska Forests, Research Unit 1652
Donald C. Schmiege, Project Leader
- Improved silvicultural practices
- Factors influencing forest diseases and insects
- Wildlife habitat and forest management interactions
- Slope stability and soil mass movement
- Wildlife habitat in coastal wetlands

Anadromous Fish Habitat, Research Unit 1705
William R. Meehan, Project Leader
- Habitat requirements of salmonids
- Effects of land use activities on fish habitat
- Anadromous fish habitat rehabilitation and enhancement

(A portion of this project is located in Corvallis, Oregon.)

Ecosystem Structure, Function, and Research

Jim Sedell and Fred Swanson

I. Erosion and ecosystem process

   A. Sediment budget for a small watershed

   B. A broad landscape view

      1. Patterns and frequency of natural disturbances

      2. Patterns and frequency of management activities

II. Forest ecosystems--lessons from old-growth

   A. Distinctive features of old-growth

      1. Big trees

      2. Dead wood

      3. Others

   B. Contrasts with natural and managed second growth

   C. The successional trade

      1. Dead wood

      2. Nitrogen fixation

III. Stream ecosystems

   A. Energy base and food wells of streams

      1. Invertebrates

      2. Fish

   B. River continuum

      1. Changes in organisms

      2. Changes in structure

      3. Changes in processes

   C. Implications for management

ADDITIONAL REFERENCES

Franklin, Jerry F.; Cromack, K. Jr.; Denison, W.; McKee, A.; Maser, C.; Sedell, J.; Swanson, F.; Juday, G. 1981. General Technical Report PNW-118. (provided)

Swanson, F. J.; Fredriksen, R. L.; McCorison, F. M. 1982. Material transfer in a Western Oregon forested watershed. In: Edmonds, Robert L., ed. Analysis of coniferous forest ecosystems in the western United States. US/IBP Synthesis Series 14. Stroudsburg, PA: Hutchinson Ross Publishing Company. (available on request)

Triska, F. J.; Sedell, J. R.; Gregory, S. V. 1982. Coniferous forest streams. In: Edmonds, Robert L., ed. Analysis of coniferous forest ecosystems in the western United States. US/IBP Synthesis Series 14. Stroudsburg, PA: Hutchinson Ross Publishing Company. (available on request).

United States
Department of
Agriculture

Forest Service

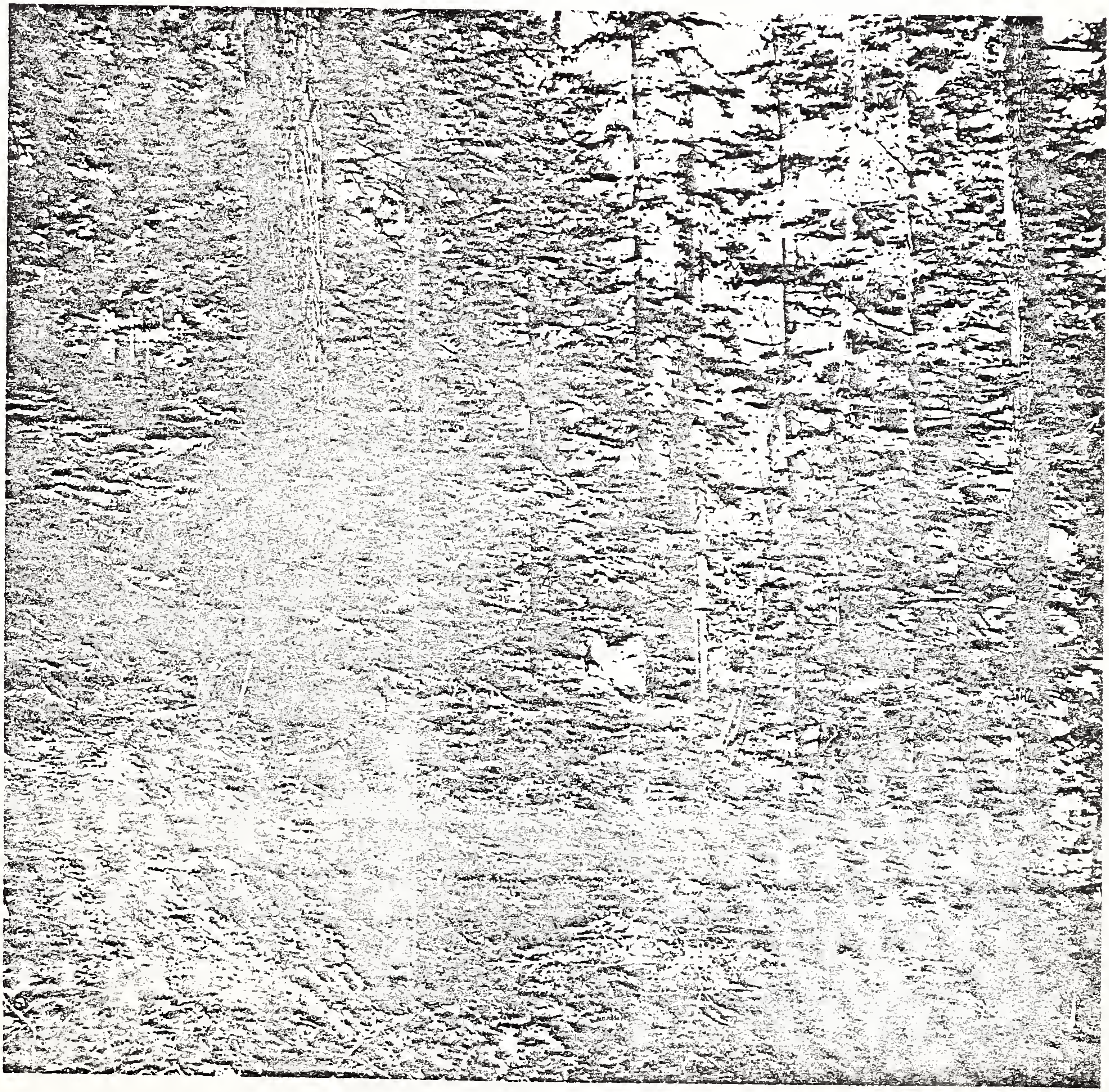
Pacific Northwest
Forest and Range
Experiment Station

General Technical
Report PNW-118

February 1981

Ecological Characteristics of Old-Growth Douglas-Fir Forests

Jerry F. Franklin, Kermit Cromack, Jr., William Denison,
Arthur McKee, Chris Maser, James Sedell, Fred Swanson,
and Glen Juday



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Summary

The major ecological features of old-growth coniferous forests in the Douglas-fir region are reviewed. Special attention is given to characteristics that distinguish old-growth forests from managed and unmanaged (natural) young stands. The primary exemplary type is 350- to 750-year-old Douglas-fir—western hemlock forest typical of the western slopes of the Cascade Range, but other types and locales are discussed. Management techniques for maintenance of old-growth forests are also considered. Major conclusions are:

1. Approximately 175 to 250 years are required to develop old-growth forests under natural conditions in both Coast and Cascade Ranges. Development of old growth is faster on good sites than on poor sites.

2. Few plant or animal species are solely confined to old-growth forests, although many species—including several vertebrates, saprophytic plants, and epiphytic lichens—find optimum habitats in such forests. Some organisms, however, may require old growth to maintain viable populations. Moreover, there are substantial differences in composition and relative abundance of species between young- and old-growth forests.

3. Gross productivity is maintained at high levels in most old-growth stands, but mortality generally balances growth. Thus, the merchantable board-foot volume tends to remain constant for several centuries or gradually decreases because the amount of defect increases. Total organic matter keeps increasing because of accumulated masses of dead tree boles, mostly as down logs.

4. Old-growth forests are highly retentive of nutrients; large amounts are incorporated into living and dead organic matter. Losses of limiting nutrients, such as nitrogen, are low.

5. Nitrogen-fixing epiphytes are abundant in old-growth trees, and bacterial nitrogen fixation appears to be common in the large woody debris characteristic of old-growth forests.

6. Small- to medium-size streams in old-growth forests depend mainly on forest litter for an energy base. These materials are invariably partially utilized before they are exported downstream.

7. The structure of old-growth forest is more heterogenous than that of young forests; coefficients of variation in tree sizes are greater, and understory patchiness is much higher than in young-growth stands.

8. Most of the distinctive features of old-growth forests can be related to four structural features: (1) large, live old-growth trees, (2) large snags, (3) large logs on land, and (4) large logs in streams. The structural features are related over time.

9. A large, old-growth Douglas-fir is individualistic and commonly has an irregularly arranged, large, coarse branch system, and often, a long crown. It is ideal habitat for specialized vertebrates, such as the red tree vole, northern spotted owl, and northern flying squirrel, as well as nitrogen-fixing lichens.

10. Large snags are valuable as habitat for a variety of vertebrates and invertebrates and as a future source of logs.

11. Logs on the forest floor are important habitats for small mammals, including species that disperse spores of mycorrhiza-forming fungi. They also are sites for substantial bacterial nitrogen fixation and are essential as seedbeds for some trees and shrubs.

12. Logs are critical to maintenance of physical and biological stability in headwater streams. Debris dams create stepped stream profiles that dissipate energy otherwise used for transporting sediment and lateral-cutting and downcutting of stream channels. Such dams, with their associated plunge pools and beds of trapped gravels and fine sediments, provide a range of habitats needed to maintain a full array of stream and stream-margin organisms. Logs are an important source of energy, and the bulk of the ntirogen supply of a stream comes from woody debris.

13. Foresters wishing to maintain or create ecosystems with old-growth characteristics can tie management schemes to maintenance or development of the four key structural components—large live, old-growth trees, large snags, and large logs on land and in streams.

14. Watersheds are probably best suited as management units for old-growth ecosystems. A small drainage usually has greater terrestrial habitat variability than occurs in a single stand, as well as a complete stream system. The size of a management unit will vary but probably should be at least 300 acres (120 hectares) to reduce effects of edges and susceptibility to damaging agents, such as wind, as well as to maintain viable populations of some birds and small mammals.

15. Buffer or leave strips along streams are also useful areas to manage as old-growth sites because woody debris is provided to the stream and the riparian zone, a particularly rich and critical wildlife habitat, is protected. Such buffers, along with roadside strips of old-growth forest, also provide migration routes for wildlife between otherwise isolated patches of mature or old-growth forest.

16. Some ecological aspects of old-growth forests can be maintained by managing for individual attributes; for example, leaving scattered old-growth trees, rotten logs, or snags on cutover lands. The linked nature of these key structural components, as well as the requirements of some organisms for the total environment of an old-growth stand, makes management of entire stands a simpler approach to retention of such ecological features.

Abstract

Franklin, Jerry F., Kermit Cromack, Jr., William Denison, Arthur McKee, Chris Maser, James Sedell, Fred Swanson, and Glen Juday.
1981. Ecological characteristics of old-growth Douglas-fir forests. USDA For. Serv. Gen. Tech. Rep. PNW-118, 48 p. Pac. Northwest For. and Range Exp. Stn., Portland, Oreg.

Old-growth coniferous forests differ significantly from young-growth forests in species composition, function (rate and paths of energy flow and nutrient and water cycling), and structure. Most differences can be related to four key structural components of old growth: large live trees, large snags, large logs on land, and large logs in streams. Foresters wishing to maintain old-growth forest ecosystems can key management schemes to these structural components.

Keywords: Ecosystems, old-growth stands, stand composition, stand structure, Douglas-fir, *Pseudotsuga menziesii*, western hemlock, *Tsuga heterophylla*.

Contents

Introduction

Old-growth forests—forest ecosystems that have developed over a long period essentially free of catastrophic (including human) disturbance—are in increasingly short supply. In the Pacific Northwest, coniferous forests dominated by large, old trees occupied large expanses of the presettlement landscape, despite periodic episodes of wildfire. These forests represented both a valuable resource (large volumes of high quality wood) and a hindrance to agricultural development. Consequently, their elimination began early and has progressed to the point where, today, most of the remaining old-growth forests are on Federal lands, including the National Forests.

At current harvest rates, old-growth stands will not be completely cut over for at least four decades, even on National Forest lands where timber production is a primary objective. And many tracts of old-growth forest are permanently protected in National Parks, Wilderness, Research Natural Areas, and similar reserves. Nevertheless, these reserves occupy less than 5 percent of the original landscape, and the end of the unreserved old-growth forests is in sight. The public, scientists, and land managers are increasingly concerned about whether species, communities, and functions are in danger of being eliminated. Are there unique features, species, or important values associated with old-growth forests? Foresters are responding to such concerns by considering longer forest rotations on some areas or preserving specimen groves within the managed forest landscape.[1]

The expanded interest in old-growth forests surfaces many unanswered questions. Clearly, an old-growth forest is more than a collection of some large, old trees; but what else characterizes these forests? How are old-growth forests distinguished from natural second-growth forests that follow fire or result from managed stands? Once key attributes of old-growth forests are defined, further practical questions remain, such as: What characteristics should be sought by foresters attempting to perpetuate or recreate such ecosystems? What size tracts are essential to maintain a viable ecological unit, and what is the best geographic distribution of areas?

A group of scientists and land managers gathered at a work session sponsored by the USDA Forest Service in February 1977 to address these questions. The objective was to identify the ecological characteristics of old-growth coniferous forests and how they differ from young-growth and/or managed forests to provide suggestions for management strategies, and to identify areas for future scientific research.

This report represents current knowledge about the characteristics of an old-growth conifer forest in the Douglas-fir region of the Pacific Northwest. Special attention is directed to management problems, selection of old-growth reserves, development of old-growth stands by long rotations, and perpetuation of attributes of old growth in areas under intensive forest management. Much more research will be necessary, however, to provide definitive guidance in such topics as distribution and necessary size of old-growth management areas.

Definitions and Assumptions

Old-growth Douglas-fir—western hemlock forests are the primary example of old-growth ecosystems in western Oregon and Washington. (Scientific names of trees are listed on page 44.) These forests, generally 350 to 750 years old, are the most common type of old-growth forest, particularly on western slopes of the Cascade Range. They are, in fact, the type of ecosystem commonly associated with the term "old growth," although forests as young as 200 years and as old as 1,000 years are also known as old growth. Other species and types also occur as old growth, such as Sitka spruce—western hemlock forests of coastal Oregon and Washington and the subalpine true fir-hemlock forests found the length of the Cascade Range. These will be commented on to the extent that available data permit.

In the strict sense, 350- to 750-year-old forests in the Douglas-fir region are generally not climax forests. Most stands of this age retain a significant component of long-lived Douglas-fir in the dominant tree canopy and will continue to do so for several more centuries. Because Douglas-fir is a subclimax species on most sites, subject to replacement by western hemlock and other more tolerant associates, stands of this type are technically in a subclimax condition. The structure and composition of the understory (as opposed to the tree canopy) are thought to be essentially the same as in a climax forest. The point is that old-growth Douglas-fir forests are not climax forests. True climax forests lack the large dominant Douglas-firs that give the forests much of their character and have tree layers of one or two shade-tolerant tree species (for example, western hemlock) which usually do not get as large. Other old-growth forests typically have long-lived pioneer species that attain large sizes, giving them much of their character, but which are generally lost in a climax forest. Examples are noble fir in subalpine environments, Sitka spruce along the coast, and sugar pine in southwestern Oregon.

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[1] As an example, the preferred management alternative in the draft environmental statement for the Hebo Planning Unit of the Siuslaw National Forest (USDA Forest Service Region 6 1977) commits 6,400 acres (2 560 ha) to maintenance of older forest. Under a 300-year rotation, approximately 3,000 acres (800 ha) of this would be maintained in the 200- to 300-year age class.

Coast redwood forests are one type that apparently do retain very large dominant specimens in a true climax condition (Franklin and Dyrness 1973). If Cupressaceae, such as western redcedar and Alaska-cedar, are present, very large dominant trees may remain in stands otherwise dominated by western hemlock or Pacific silver fir for yet another millenium beyond the disappearance of the Douglas-fir; members of this family have extremely long lifespans and some ability to reproduce under closed forest conditions.

In this report, the stream component of an old-growth ecosystem is considered along with the terrestrial component. Stream conditions depend strongly on the nature of associated forests. In fact, streams in old-growth forests contain several of the most distinctive features of these ecosystems.

Attributes of forest ecosystems are composition, function, and structure. Composition refers primarily to the array of plant and animal species present in an ecosystem. We also considered dominance an element of composition; that is, shifts in abundance as well as presence or absence of species. Function refers to how various ecological processes, such as production of organic matter and cycling of nutrients (through pathways and compartments), are accomplished and the rates at which they occur. It is well known that, as ecosystems develop or age, the relative size of various compartments, complexity of pathways, and transfer rates change (Odum 1971). Increased complexity in routing dead organic material (development of a system based on detritus) is one characteristic example of a functional change with succession. We considered types and rates of ecological processes as functional features of an ecosystem. Structure refers to the spatial arrangement of various components of the ecosystem, such as heights of various canopy levels and spacing of trees.

Finally, we define some terms and determine age limits for old-growth forests. It is difficult to define a lower age limit for old growth. The transition from mature to old-growth forest is gradual, and the age limit varies with site conditions (trees take different times to reach comparable size or physiological age on different sites) and with the type of stand (initial density and composition) that develops after the last disturbance. We would expect earlier development of old-growth characteristics on better Douglas-fir sites, in coastal environments, and in understocked stands and delayed development on poor sites, on many (but not all) subalpine environments, and in overstocked stands.

Forests typically begin exhibiting old-growth characteristics at about 175 to 250 years. Forests up to about 75 to 100 years old can generally be considered ecologically young in the Douglas-fir region. This is the period of very rapid growth or "adolescence." Mature forests are those in the period between culmination of maximum growth (peak of the growth curve) and the development of old-growth characteristics; that is, generally between 100 and 200 years. Federal foresters normally select the culmination of mean annual wood increment (end of youth) as the rotation or cutting age in managed forests. Substantial net growth (or net accumulation of live biomass) does continue in the mature forest although at a slower rate than in the young stand.[2] There is typically little net gain or loss of live biomass in most old-growth forests over the long run, barring some catastrophe. This may change in extremely old forests (>750 years), but few examples of such old forests exist, even in the Pacific Northwest.

In this report, we distinguish between natural young-growth forests (such as those that have followed wildfires) and managed young-growth forests. Contrasts are very sharp between natural and managed young growth in several ecological attributes, such as the number of snags and rotten logs. Our concept of managed stands is based on current USDA Forest Service plans of prompt establishment of fully stocked young conifer forests after cutting of natural stands and major efforts to dispose of residue. We assume that such forests will typically have moderate levels of management (for example, thinning) and will have an 80- to 90-year rotation—a somewhat more conservative strategy than is currently practiced on the most intensively managed industrial forest lands in the Pacific Northwest.

We have attempted to contrast the features of old-growth and second-growth forests. Surprisingly, some aspects of younger forests have not yet been studied, so comparable data are not always available. In such cases, we have described the old-growth condition; future research on young growth should provide comparisons. When reading this report or drawing inferences from it, the reader is cautioned to be careful in distinguishing natural second- or young-growth stands from managed young-growth forests in any comparisons with old-growth forests.

[2] Williamson and Price (1971) show some very high rates of growth in stands at or beyond average current rotation ages. Seven plots in unthinned Douglas-fir stands 90 to 150 years of age averaged a net growth of 114 ft^3/acre (690 board feet (fbm) per acre or 8 m^3/ha) per year.

Characteristics of Old-Growth Forests

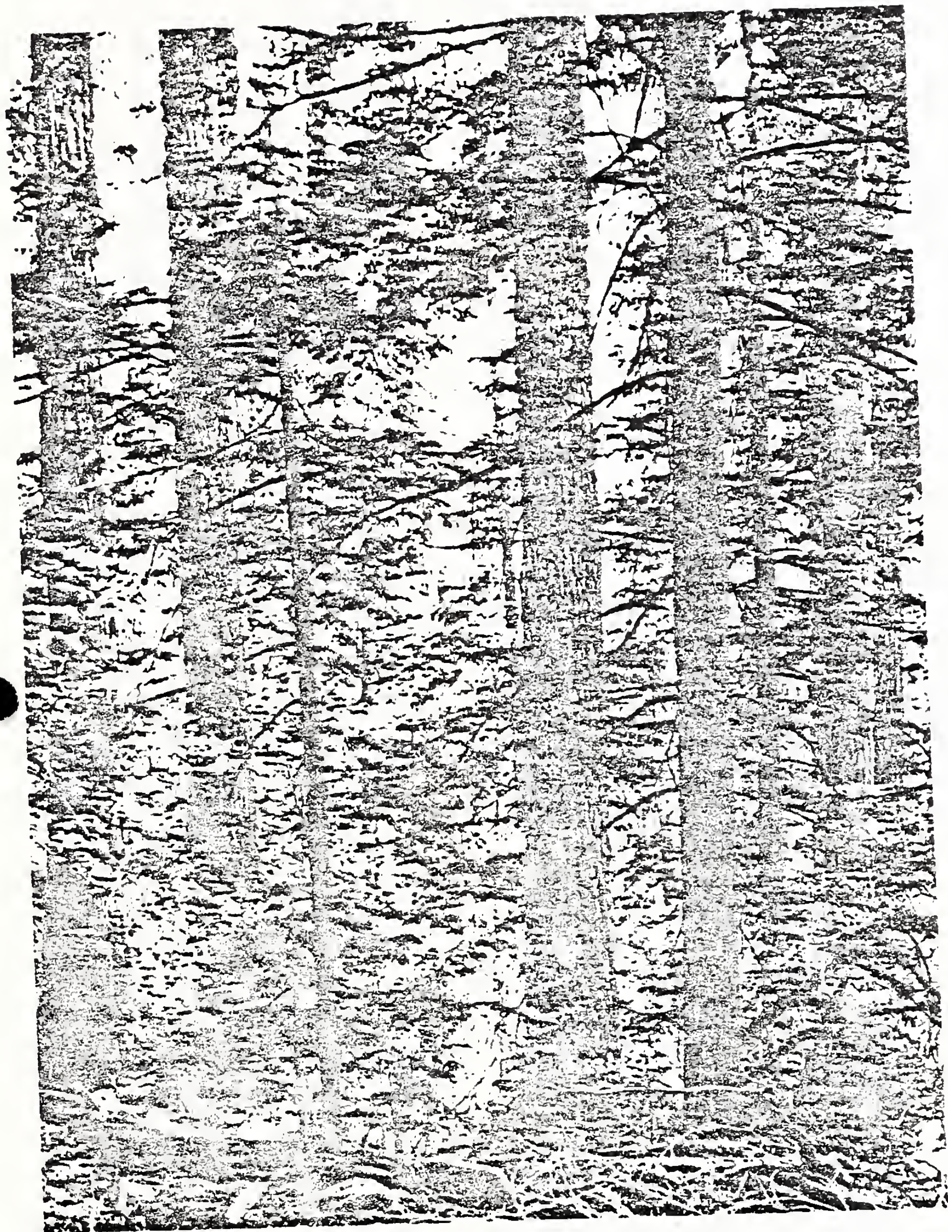

Figure 1. Old-growth Douglas-fir—western hemlock forest showing diversity of tree sizes and heterogeneity of understory.

Some general attributes of an old-growth forest are immediately apparent to an observer with even a moderate background in natural history (fig. 1). Trees typically vary in species and size; dominant specimens are truly impressive. Some large species differ in color and texture as well as in size. The multilayer canopy produces a heavily filtered light, and the feeling of shade is accentuated by shafts of sunlight on clear days. The understory of shrubs, herbs, and tree seedlings is often moderate and is almost always patchy in distribution and abundance. Numerous logs, often large and in various stages of decay, litter the forest floor, creating some travel routes for wildlife and blocking others. Standing dead trees, snags, and rotted stubs are common, although a visitor gazing toward the ground will often mistake dead trees in early stages of decay for live trees. It is quiet; few birds or mammals are seen or heard except perhaps the melody of a winter wren (*Troglodytes troglodytes*), the faint songs of golden-crowned kinglets (*Regulus satrapa*) in the tree canopies, or a chickaree (*Tamiasciurus douglasi*).

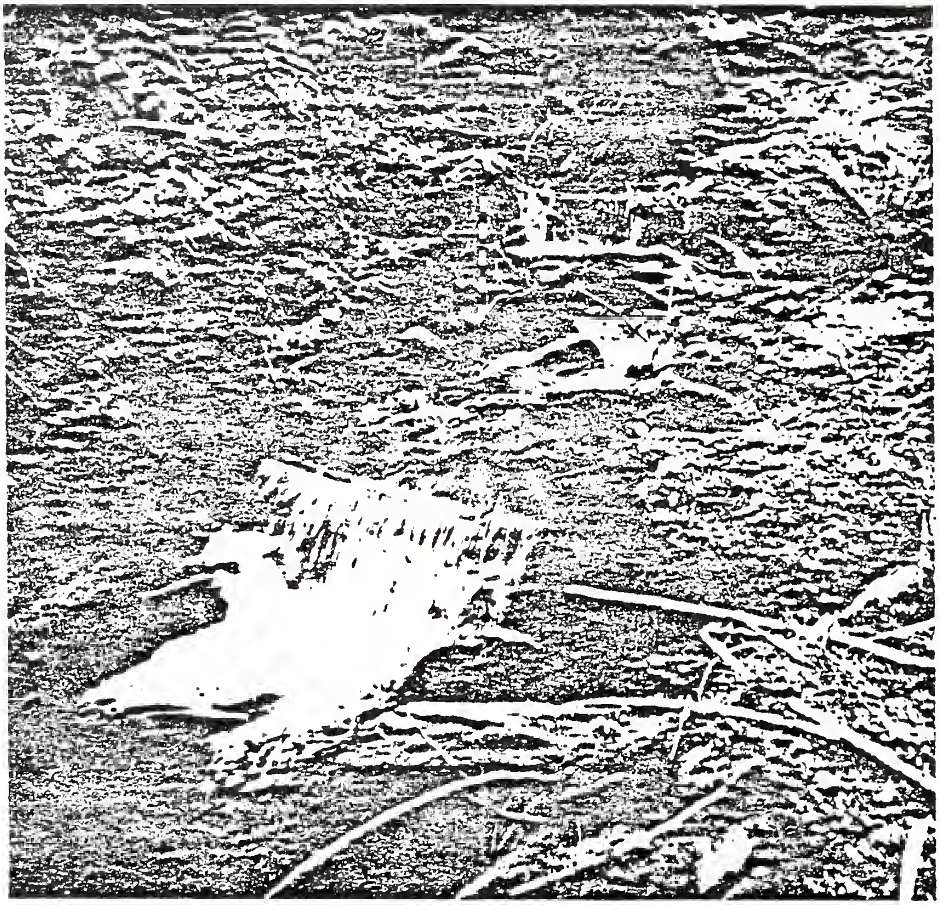

Figure 2. Small streams within old-growth forests depend heavily on terrestrial vegetation for energy and physical integrity.

Small to moderate size streams flowing through old growth (fig. 2) are shaded, often completely shielded from the sun by the canopies of adjacent trees. The smallest streams may be choked with organic debris; as size and volume of streams increase, clear, cool water runs through gravel beds behind old log dams and spills into plunge pools. Organic debris—for example, leaves, needles, twigs, bud scales—floats on the surface and accumulates in backwaters.

Some of these impressions represent important, distinctive aspects of an old-growth forest ecosystem that we will discuss as composition, function, and structure. Structural aspects of old-growth forests are the major unifying element since the peculiar compositional and functional features are mainly related to the distinctive structure of old growth. We discuss composition and function first, however, so that these aspects are not completely overshadowed by that of old-growth structure. We discuss composition and function again in the sections on habitat and cycling roles of live old-growth trees, standing dead trees or snags, logs on land, and logs in streams.

Composition

Old-growth forests obviously differ in composition from young stands. Ecological succession produces changes in the array of plant and animal species as well as in their relative abundance. Hence, there is a change from pure or nearly pure young forests of Douglas-fir to mixtures of old-growth Douglas-fir, western hemlock, western redcedar, and other species. Thomas et al. (1979c) outlined the changes in animal species associated with plant community successional stages in the Blue Mountains of eastern Oregon. These principles apply equally to Douglas-fir–western hemlock forests, as shown by Gashwiler (1970) and others. The most sterile successional stage, in diversity of both plant and animal species, is a dense, rapidly growing young conifer forest (Edgerton and Thomas 1978, Long 1977, Meslow 1978).

Few vascular plants appear confined to old-growth ecosystems in the Douglas-fir region. Lists of species from old-growth Douglas-fir–western hemlock stands in the H.J. Andrews Experimental Forest (Dyrness et al. 1974) show none that are confined to old-growth forests.

Some vascular plants do find optimum habitat (or most frequently—suitable environments) in old-growth Douglas-fir ecosystems. These are often saprophytic[3] plants belonging to the orchid and heather families—for example, phantom orchids (*Cephalanthera austinae*), pinesap (*Monotropa hypopitys* L.), woodland pinedrops (*Pterospora andromeda* Nutt.), and candystick (*Allotropa virgata*)—which favor heavily shaded environments rich in organic debris. Again, these vascular plants are not confined to old-growth ecosystems but often find suitable environments there.

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[3] Saprophytic plants obtain all or part of their energy from decomposition of dead organic materials rather than by photosynthesis. Most vascular plants characterized as saprophytic have fungal associates essential to their survival (Furman and Trappe 1971).

For lower plants—including mosses, lichens, liverworts, algae, and bacteria—there currently is no way of systematically addressing the question of dependency of a species on old growth. Many species find optimum habitat in old-growth forests, and some probably require old-growth habitat for survival. As in the case of vascular plants, more species probably find their optimum habitat in old growth, however. For example, snags in old-growth forests have a rich flora of Caliciales (lichens). Species that occur in old growth rarely occur on the drier snags found in younger forests. The rich communities of epiphytes found in the canopies of old-growth Douglas-fir include the foliose lichen *Lobaria oregana* (seldom found elsewhere), as well as *Lobaria pulmonaria* and *Alectoria sarmentosa* (also found in younger stands).

More is known about the relationship of vertebrate animals to old growth than any other group of organisms. Much of the interest in old growth is as habitat for the vertebrates primarily found there (Thomas 1979, DeGraaf 1978). Although many vertebrates utilize old-growth forests, some (table 1) exist mainly in old-growth ecosystems. The degree to which the listed species depend on old-growth forest ecosystems varies, but all find optimum breeding or foraging habitat there. Whether any species depends totally on old growth for survival is not clear; however, the fact that a species can survive in other age classes of a forest does not necessarily mean it can survive once the major reservoir of optimum habitat is gone. The factors responsible for a species' orientation toward old growth (food source, suitable nesting sites, protection from competition) also vary between species and provide a key to the management of a particular species (for example, providing snags for cavity dwellers).[4]

Table 1—Vertebrate animals that find optimum habitat for foraging or nesting or both in old-growth Douglas-fir–western hemlock forest ecosystems[1]

Group	Common name	Scientific name
Birds	Goshawk	*Accipiter gentilis*
	Northern spotted owl	*Strix occidentalis*
	Vaux's swift	*Chaetura vauxi*
	Pileated woodpecker	*Dryocopus pileatus*
	Hammond's flycatcher	*Empidonax hammondii*
	Pine grosbeak	*Pinicola enucleator*
	Townsend's warbler	*Dendroica townsendi*
Canopy mammals	Silver-haired bat	*Lasionycteris noctivagans*
	Long-eared myotis	*Myotis evotis*
	Long-legged myotis	*Myotis volans*
	Hoary bat	*Lasiurus cinereus*
	Red tree vole	*Arborimus longicaudus*
	Northern flying squirrel	*Glaucomys sabrinus*
Ground mammals	California red-backed vole	*Clethrionomys californicus*
	Coast mole	*Scapanus orarius*
	Marten	*Martes americana*

[1] On the Olympic Peninsula, fisher (*Martes pennanti*) exists only in old-growth forests and should be added to this list for at least the peninsula. Habitat changes caused by cutting, rather than trapping, are probably responsible for elimination of this species from other sites. (Personal communication from Bruce Moorhead, Research Biologist, Olympic National Park, May 16, 1978.)

[4] Meslow (1978) reported that of 84 birds found in Douglas-fir forests, 49 nest in mature stands, and 3 of those (goshawk, northern spotted owl, and Vaux's swift) nest almost solely in mature stands. Sixteen of 20 hole nesters are found in mature forests.

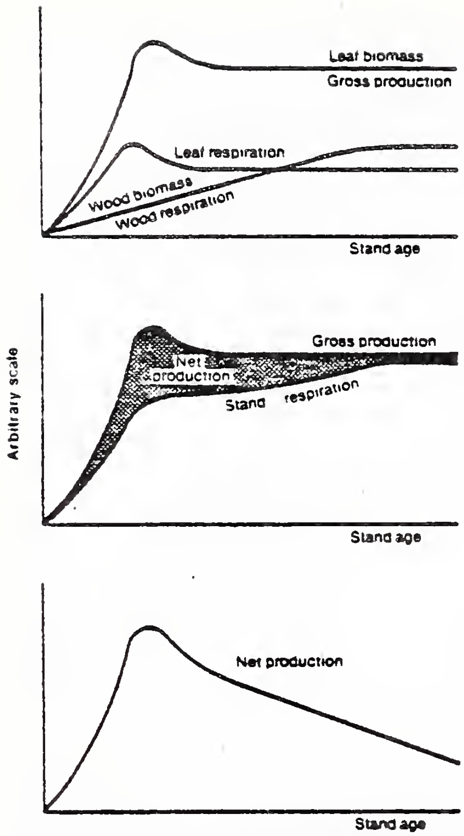


Figure 3. Trends in biomass, respiration, and production in forest stands; note increasing amounts of energy used in respiration (based on Shidei and Kira 1977).

Primary ecological functions within a forest ecosystem include primary production, energy flow, conservation and cycling of nutrients, and regulation of waterflow.

Forests. Primary production in old-growth forests is typically high. Leaf area and biomass accumulate rapidly and stabilize fairly early in the life of a stand (Long and Turner 1975), but there is little evidence for a substantial decline in either in old-growth stands. Foliage biomass values of 6.2 to 11.6 tons per acre (14 to 26 tonnes/ha) and ratios of projected leaf areas per unit of ground area ranging from 7 to 16 were reported in 14 old-growth Douglas-fir forests (Franklin and Waring 1980); leaf areas were affected by environmental conditions (Gholz et al. 1976). Values for coastal and subalpine types of old growth are comparable. These values exceed those found in mature (second-growth) forests 90 to 130 years in age. The large leaf areas in many old-growth stands indicate shade-tolerant associates, such as western hemlock. The long-crowned Douglas-firs also contribute substantial leaf area, however, and the multilayered canopy of an old-growth forest is well suited to efficient capture of energy. In summary, the "factory" for photosynthetic production is generally large and intact in an old-growth forest.

Production of photosynthate may be high in mature and old-growth stands, but total energy used in respiration is also high. A hypothetical example of trends in biomass, respiration, and production rates (fig. 3) shows the increasing energy cost (in respiration) of simply maintaining, as living tissue, the increasing accumulations of biomass. Grier and Logan (1977) estimated gross respiration in one old-growth forest to be 67 tons per acre (150 tonnes/ha) per year—over 90 percent of the gross production.

On a more practical level, substantial growth in the form of wood and other increments of biomass occurs in old-growth forests. Much, but not all, of this increment is on the western hemlocks, western redcedars, and other shade-tolerant associates. But little or no increase of additional living biomass—or cubic feet—is occurring in an old-growth stand because of mortality and, in some stands, disease. Over the long run, living biomass appears to fluctuate around a plateau in response to episodes of heavy and light mortality.

This pattern of subtantial growth or increment in old-growth stands which is largely offset by mortality has been documented by the few available studies of growth. Annual wood increment was 1,582 board feet or 226 cubic feet per acre (15.8 m^3/ha) over a 10-year period in a 250-year-old Douglas-fir stand in the Clackamas River drainage of Oregon (Berntsen 1960). The annual mortality of 1,156 board feet or 201 cubic feet per acre (14.1 m^3/ha) did not completely offset this large amount of growth even though the measurement period was one of heavy loss; over 15 percent of the original 48 Douglas-firs per acre (119/ha) were lost in the 10-year period, mainly to bark beetles and windthrow. An old-growth[5] Douglas-fir–western hemlock forest in the Wind River drainage of the southern Washington Cascade Range grew 699 board feet or 106 cubic feet per acre (7.4 m^3/ha) per year over a 12-year period.

[5] King (1961) indicated that the age of the stand was 350 years. Subsequent stump counts in adjacent clearcuts, which had comparable forests before the cutting, suggest that the stand originated closer to 450 years ago.

Epidemics of bark beetles in Douglas-fir and western white pine were major contributors to the average annual mortality of 614 board feet or 96 cubic feet per acre (6.7 m³/ha). Despite these large losses, the stand registered a small net gain. Neither Berntsen (1960) nor King (1961) took into account losses from decay in living trees, so some further reduction from the net growth figures is necessary in calculating net production of sound wood. Grier and Logan (1977) report losses in aboveground biomass in trees in a 25-acre (10-ha), 450-year-old stand because of high mortality in a 2-year study period.

Old-growth forests, as well as much younger ones, apparently do experience periods of high mortality from bark beetle epidemics, severe windstorms, and so forth. There is also evidence that infection and losses caused by heart rots (and perhaps other types of diseases) are cyclic (Boyce and Bruce Wagg 1953). The limited data available indicate, however, that neither leaf mass nor total live biomass show regular reductions with age, at least up to 600 or 700 years.

Regardless of whether the volume of living trees is level or declining, there is evidence that total organic matter—living and dead—continues to increase in old-growth forests. This is mainly in the form of coarse woody debris, such as logs, which accumulates much more rapidly than it decomposes. In the absence of wildfire or other major disruption, the ultimate consequences of this continuing accumulation of organic matter are not clear; few stands are old enough to indicate when accumulations of organic matter will peak, and even fewer have been analyzed. Limited data suggest that total accumulations of organic matter may peak at 800 to 1,000 years (Franklin and Waring 1980).

Development of large accumulations of dead organic matter relates to a major distinction in nutrient cycling between young-growth and old-growth stands. Specifically, Odum (1971) refers to carbon and nutrient cycling in old-growth forests as a "closed cycle, detrital-based system." Nutrient paths are complex and involve large amounts of detritus. Detritus decomposes slowly through the efforts of heterotrophic organisms; for example, fungi, bacteria, and invertebrates. Although release of energy and nutrients from the dead organic materials is slow, ecosystems functioning in this way are extremely conservative—nutrients are tightly retained within the system (Fredriksen 1970, 1972; Fredriksen et al. 1975). The low level of nutrients and other dissolved and suspended materials in water from old-growth ecosystems reflects this conservatism. The low level of erosion and the nature of the streams also contribute to the low losses of nutrients from old-growth forests (Sollins et al. 1980).

The development of a strongly conservative carbon and nutrient cycle based on detritus is a gradual and continuing process from the time a forest becomes established until its ultimate destruction by a natural catastrophe or human intervention; such features are not solely confined to old-growth forest. Old-growth forests usually have greater amounts and larger sizes of dead organic matter than young and mature stands of natural origin, however, and much more than managed forests.

In addition to being highly retentive of nutrients, old-growth forests have other means that provide nutrients as more and more nutrients become bound in wood and other materials. Trees meet larger and larger proportions of their annual requirements of nutrients by internal redistribution (Waring and Franklin 1979). Nitrogen (N) is a particularly critical nutrient; in an old-growth forest, substantial annual increments of N are provided by foliose lichens in tree crowns and by bacterial fixation in coarse woody debris. These biological fixation processes may provide inputs of 6 to 9 pounds of N per acre (7.5 to 9.5 kg/ha) or more per year to old-growth systems—more than in natural young growth and, especially, intensively managed forests.

The net effects of hydrologic cycling in old-growth and younger forests are similar. Water usage by plants and, therefore, streamflow quickly return to near the original level (of the natural forest) after a cutting (Anderson et al. 1976). There may be minor contrasts between young growth and old growth in that deep crowns of the old growth may have some advantage in intercepting fog and other aerosols. Perhaps the major hydrologic contrasts between young and old forests are in effects on erosion and on the biological and physical natures of streams.

Table 2—Annual transfer of inorganic and organic matter under old growth on watershed 10 to a channel by hill slope processes, and export from the channel by channel processes, H. J. Andrews Experimental Forest, western Oregon Cascade Range[1]

Process	Inorganic matter		Organic matter	
	Tonnes	Tons	Tonnes	Tons
Hill slope:				
Solution transfer	3.0	3.3	0.3	0.3
Litter fall	0	0	.3	.3
Surface erosion	.5	.6	.3	.3
Soil creep	1.1	1.2	.04	.04
Root throw	.1	.1	.1	.1
Debris avalanche	6.0	6.6	.4	.4
Slump and earthflow	0	0	0	0
Total	10.7	11.8	1.4	1.4
Channel:				
Solution transfer	3.0	3.3	.3	.3
Suspended sediment—				
Gross	.8	.9	.1	.1
Net	.6	.7	.1	.1
Bedload	.6	.7	.3	.3
Debris torrent	4.6	5.1	.3	.3
Total	8.9	10.7	1.0	1.0

[1] Adapted from Swanson et al. (1981a).

A broad spectrum of geomorphic processes transports organic and inorganic materials down hill slopes and through stream channels in old-growth ecosystems. Swanson et al. (1981a) estimate rates of material transfer for seven hill slope and four channel processes in one old-growth watershed (table 2). Surprisingly, greatest transfer of inorganic material occurs by the most episodic processes—debris avalanches and torrents from hill slopes and down stream channels. For example, only one debris avalanche moving more than 2,625 cubic feet (75 m^3) of soil is estimated to occur in three or four centuries of forested conditions in this steep, 25-acre (10-ha) watershed. Yet, this process moves more material, averaged on an annual basis, from hill slopes than more persistent and pervasive processes, such as surface erosion and solution transfer. Comparable data for second-growth forests are not available, but Swanson et al. (1981a) review and discuss variations in importance of erosion processes during the first 10 to 15 years after clearcutting.

The most important physical influence of old-growth forests on erosion is the provision of large organic debris which slows the routing of sediment through the channel system and dissipates the energy of streams.

Streams. The food base or energy supply of a stream in an old-growth stand is mostly litter from the adjacent forest—leaves, needles, cones, twigs, wood, and bark, all known as allochthonous material to the aquatic biologist (table 3). Large boles require special consideration since they (1) enter streams infrequently rather than annually, (2) physically shape the small stream, and (3) move downstream only by the rare debris torrent and must, therefore, be biologically processed or broken down in place. Input of allochthonous material to streams and their export from streams are predictable. Of the organic material that falls or slides into small streams each year, only 18 to 35 percent is exported or

Table 3—Particulate organic budget of a small stream of the H. J. Andrews Experimental Forest in the western Oregon Cascade Range, watershed 10, 1973-74 water year

Input			Standing crop			Export		
Source	Amount per stream per year		Source	Amount per stream per year		Source	Amount per stream per year	
	Pounds	Kilograms		Pounds	Kilograms		Pounds	Kilograms
Lateral movement[1]	428	194	Large detritus[2]	9,984	4 530	Dissolved organic matter	408	185
Dissolved organic matter	408	185	Small detritus[2]	1,587	720	Particulate organic matter	372	169
Litter fall and throughfall	251	114	Fine detritus[2]	115	52	Detritus respiration	205	93
Moss primary production	15	7	Primary producers	26	12	Primary producer respiration	33	15
Algal primary production	2	1	Macroinvertebrates	1	.3	Macroinvertebrate respiration	4	2
Total	1,104	501	Total	11,713	5 314	Total	1,022	464

[1] Lateral movement refers to materials sliding downslope into streams.
[2] Large detritus = >4 inches (10 cm); small = 0.4 inch to 4 inches (1 to 10 cm); fine = <0.4 inch (1 cm).

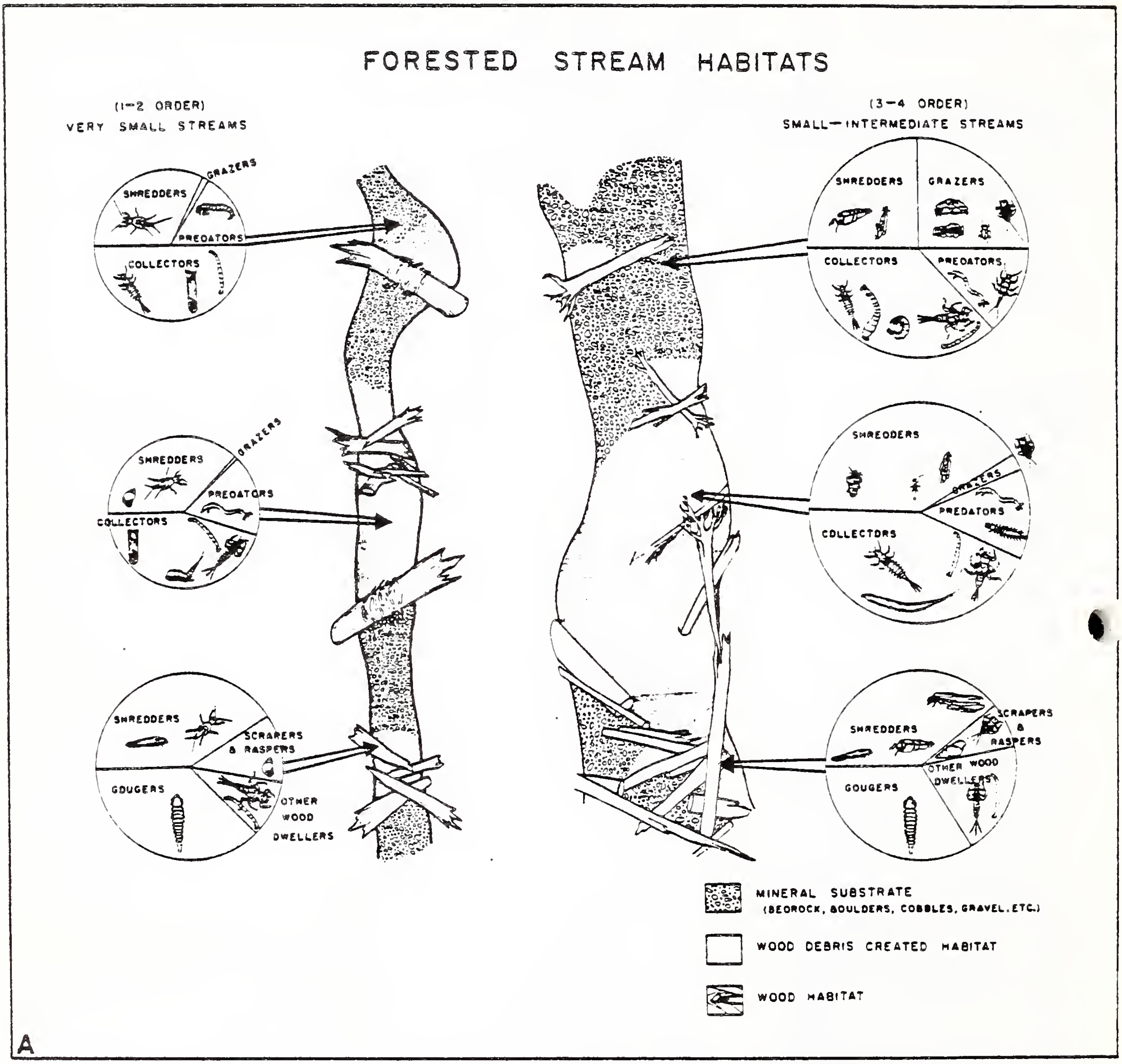
flushed downstream. These streams under old-growth forests retain materials; they are not mere conduits for quick export from the system—60 to 70 percent of the annual organic inputs are retained long enough to be biologically used by stream organisms. Large dams of woody debris effectively act as sieves and deposit zones for fine organic matter, allowing time for microbial colonization and consumption of the material by insects.

The amount of various inputs of debris processed in a defined section or "reach" of a stream depends on (1) the quantity and quality of nutrients in the debris and (2) the capacity of the stream to retain finely divided debris for the time required to complete processing.

Debris may be fully utilized by the biotic community within a reach of stream or may be exported downstream. Since export from one reach constitutes an input to downstream reaches, processing continues as small debris moves through the system. Processing of organic matter includes metabolic utilization by bacteria and fungi, consumption of debris by insects, and physical abrasion. In all cases, debris is broken into smaller pieces which increases the surface-to-volume ratio and makes the debris particles increasingly susceptible to microbial attack. The length of stream required for complete processing of organic material may be longer in years when water is high than when it is low; the same is true in streams with lower capacity for retaining fine debris, such as streams lacking sufficient debris dams.

The point of this discussion is that small first- and second-order streams feed larger streams with a partially prepared food resource.[6] The stream is a continuum in which transported food materials become progressively smaller. In small streams under old growth, a large proportion of the basic food resource is derived from wood.

[6] As defined by Strahler (1957), first-order streams are smallest streams with no tributaries; junctions of first-order stream segments mark the upstream end of a second-order stream segment; the junction of two second-order segments marks the head of a third-order segment, and so on.


FORESTED STREAM HABITATS
(1-2 ORDER)
VERY SMALL STREAMS
(3-4 ORDER)
SMALL-INTERMEDIATE STREAMS
SHREDDERS
GRAZERS
PREDATORS
COLLECTORS
SCRAPERS & RASPERS
GOUGERS
OTHER WOOD DWELLERS
MINERAL SUBSTRATE
(BEDROCK, BOULDERS, COBBLES, GRAVEL, ETC.)
WOOD DEBRIS CREATED HABITAT
WOOD HABITAT
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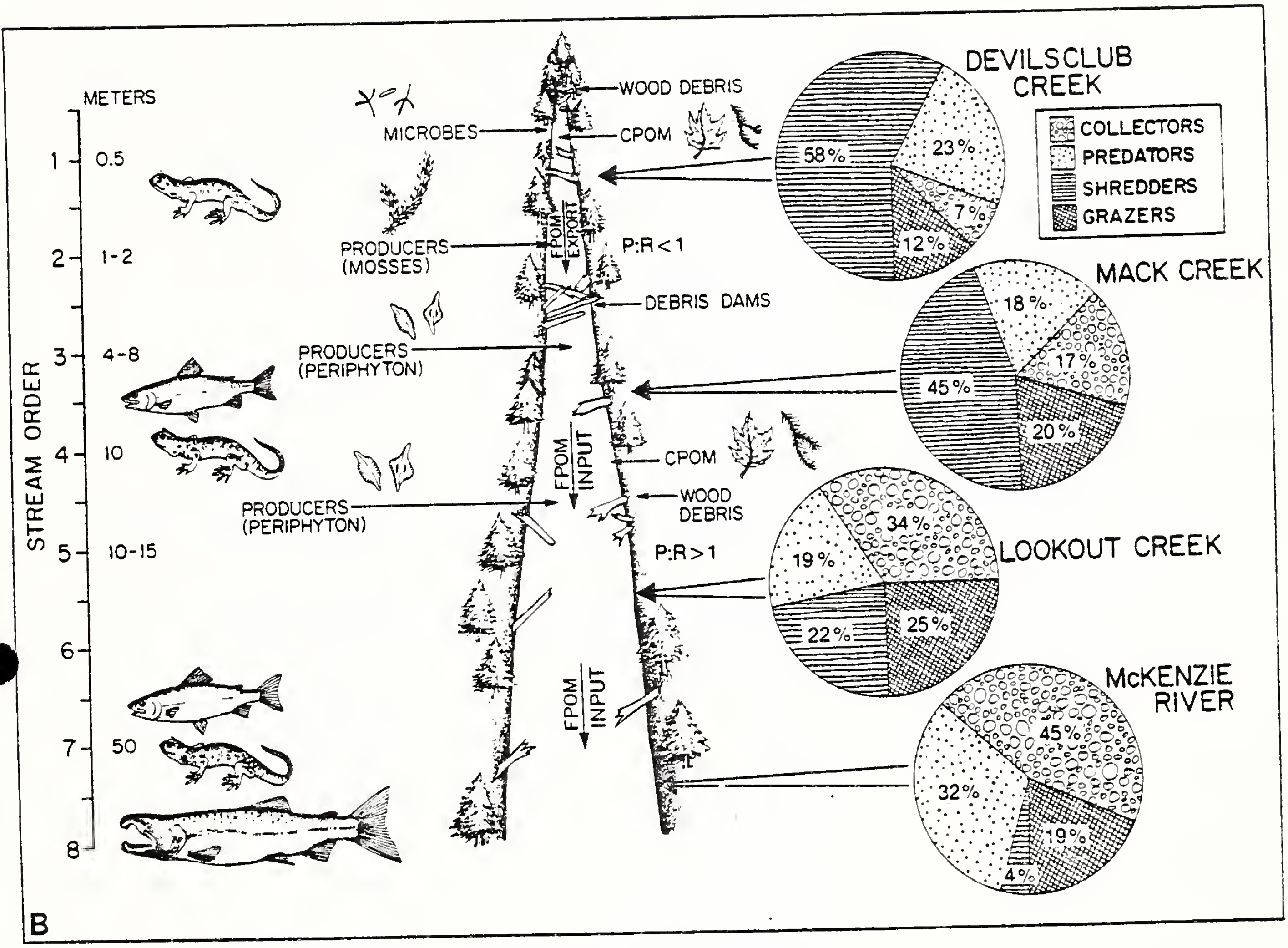


Figure 4. Streams in old-growth forests have a rich, invertebrate fauna which gouge, shred, and scrape the larger organic material and collect the fine organics carried in the water. The type of fauna shifts with size of stream, however, and third- to fourth-order streams are generally richer in numbers, biomass, species, and functional types than are smaller streams.
A. Comparison of very small and intermediate streams, showing contrasting proportions of various functional groups by substrate (Anderson and Sedell 1979).
B. Diagram of first- to eighth-order streams, showing width (meters), dominant predators, producer groups, P:R (production:respiration) ratios, importance of wood, and proportion of invertebrate functional groups (adapted from Anderson and Sedell 1979). CPOM = coarse particulate organic matter; FPOM = fine particulate organic matter.

The influence of the forest diminishes as the stream gets larger. The energy base of the stream is derived more from algae and less from forest litter (fig. 4, A and B). The greatest influence of the forest is found in the very small streams, whereas the most diversity in inputs and habitats is found in the intermediate (third- to fifth-order) streams. The invertebrates reflect these downstream shifts with fewer shredders (leaf eaters) and more grazers (algal feeders) in the larger streams (fig. 4, A and B). As the size of a stream in an old-growth forest changes, there are shifts in dominant organisms and the role each group of organisms plays in using available organic materials (fig. 4, A and B).

Table 4—Inputs of litter (excluding bole wood) to a small stream in watershed 10 in old-growth Douglas-fir, H. J. Andrews Experimental Forest, western Oregon Cascade Range[1]

Type of litter	Weight		Percent of total
	Ounce per square foot	Grams per square meter	
Leaf:			
Hardwood	0.0005	0.166	7.6
Conifer	.0016	.489	22.5
Fine woody litter (<1-centimeter or 0.4-inch diameter)	.0032	.972	44.7
Coarse woody litter (1- to 10-centimeter or 0.4- to 4-inch diameter)	.0018	.548	25.2

[1] Woody debris totals nearly 70 percent of the litter, even though the large weight of wood greater than 10 centimeters in diameter is excluded.

Table 5—Input of particulate organic nitrogen to a small stream in old-growth forest (watershed 10), H. J. Andrews Experimental Forest, western Oregon Cascade Range

	Amount per year			
Source	Ounce per square foot	Gram per square meter	Pounds	Kilogram
Deciduous leaf litter	0.0014	0.44	0.68	0.31
Coniferous leaf litter	.0030	.91	1.41	.64
Fine woody litter (<1-centimeter or 0.4-inch diameter)	.0023	.71	1.10	.50
Coarse woody litter (1- to 10-centimeter or 0.4- to 4-inch diameter)	.0003	.08	.13	.06
Nitrogen fixation in wood less than 10-centimeter or 4-inch diameter	.0023	.70	1.07	.49

The percent of each type of food available to microbes and invertebrates in small streams under old growth is presented in table 4. Woody material constitutes 50 to 70 percent of the total organic material, including very fine particles derived almost exclusively from bole wood; these data do not include bole wood, however, because it would completely overwhelm the other categories in table 4.

Invertebrates in the smallest streams flowing through old-growth forests have evolved to gouge, shred, and scrape wood and leaves and to gather fine organic particles. These first- and second-order streams are loaded with wood and have many wood-gouging beetle larvae and leaf-shredding stoneflies and snails. The small particles of organic material trapped by the large wood are gathered and fed on by a benthic copepod. These streams are noted for their uniqueness in that each is predominantly a beetle-stonefly-copepod-miniature snail invertebrate community (fig. 4A), not for an abundance of invertebrate populations.

Third- to fourth-order streams are generally richer in kinds, numbers, and biomass of organisms than are smaller streams—including a rich variety of insects and a continuous population of vertebrates, such as cutthroat trout (*Salmo clarki*), tailed frog, and Pacific giant salamander. The richness and abundance are due, in part, to the increased importance of algae as a source of energy.

The green plants or primary producers in streams also vary widely by size of stream. In first- and second-order streams, moss cover is generally greater than 20 percent of the stream area and is located primarily on wood, bedrock, and boulders. The moss community generally occupies 5 percent or less of the stream area in third- and fourth-order streams—mostly on wood, bedrock, and large boulders. The algal community—primarily diatoms, green algae, and a few blue-green algae—is well developed and widely spread throughout larger streams. Small streams have a sparse diatomaceous flora and a patchy blue-green algal community which is intimately associated with the mosses.

Large woody debris is responsible for two types of habitat within each stream—wood and wood-created environments, such as depositional pools. Each of these habitats, as well as those not related to wood (mineral sediments in streambed formations not created by wood or bedrock), has a different faunal composition. Relative proportions of wood-related and other habitats vary markedly with size of streams. In the smallest streams, 50 percent or more of the area may be occupied by wood and wood-related habitats compared with 25 percent for third- and fourth-order streams.

Coarse woody debris also functions as a major source of N. Although the content of N in wood is small, large amounts of N are fixed in and on woody debris. In one small watershed in the western Oregon Cascade Range, input of N in wood or fixed in wood accounts for 52 percent of the total input of N to the stream, not counting N in woody material over 4 inches (10 cm) in diameter (table 5).

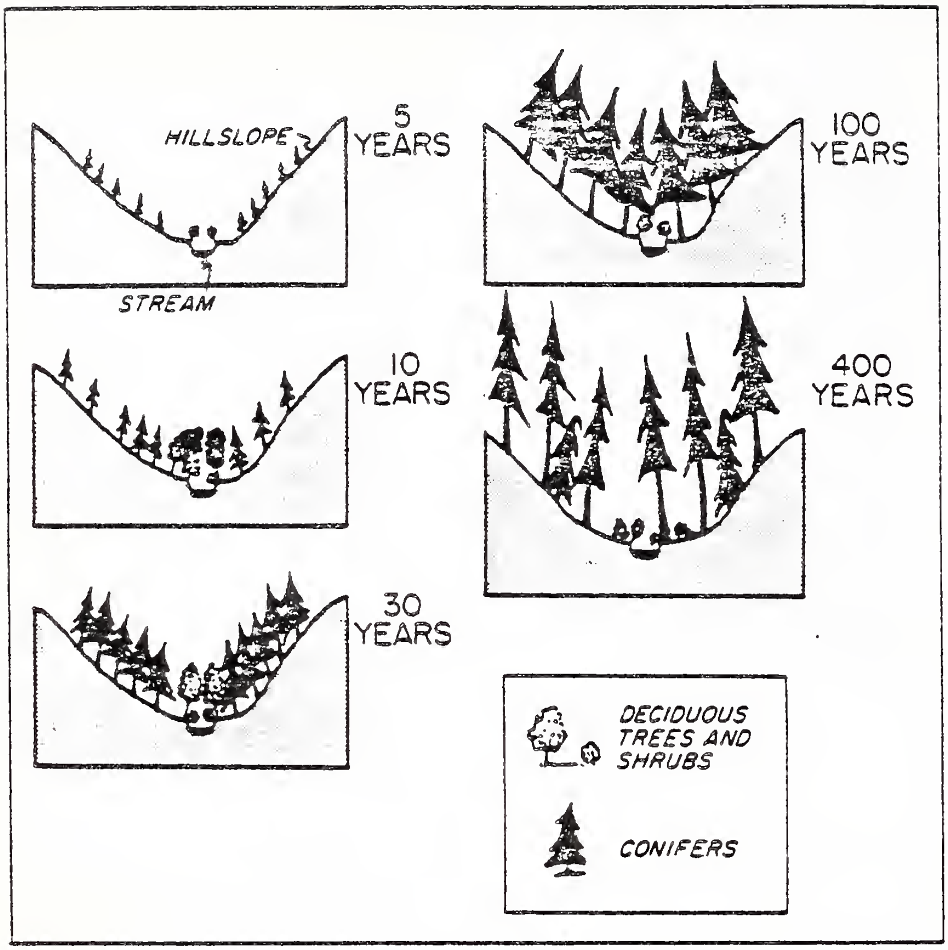


Figure 5. Hypothetical changes in the riparian zone through succession in Douglas-fir—western hemlock forests (Swanson et al. 1981b; from Analysis of Coniferous Forest Ecosystems in the Western United States *US/IBP Synthesis Series, V. 14, edited by R. L. Edmunds. Copyright © 1981 by Dowden, Hutchinson & Ross, Inc., Stroudsburg, Pa. Reprinted by permission of the publisher.)*

The environment of streams dominated by forest varies with stand age in response to changes in structure of streamside vegetation as it develops through time. A hypothetical succession of streamside vegetation is shown in figure 5. In small streams in the western Cascade Range, dense riparian stands of deciduous shrubs and small trees develop in one to two decades. Beyond that point in development, upslope conifers begin to overtop the deciduous streamside vegetation. Stands 50 to several hundred years old have a dense canopy of deciduous trees with little understory vegetation. Older stands have a multilayered structure, and more light penetrates to streams.

These variations in structure of the stand through time are reflected by shifts in both energy base and habitat in streams up to about fourth order. Although little is known about actual productivity of streams through succeeding stand structures, algae are known to be a dominant source of energy before the canopy closes; they continue to be an important contributor in small to intermediate streams as long as there is a hardwood canopy cover. When the stream is completely enclosed by a conifer canopy, the ecosystem shifts to a food base of conifer litter which is of lower quality than algae. The more open canopy of old growth provides greater diversity of nutrient inputs, including algae and litter of herbs, shrubs, hardwoods, and conifers.

Murphy (1979) measured populations of vertebrate and invertebrate predators, including salamanders in streams in recent clearcuts (less than 10 years old), in 20- to 30-year-old, second-growth stands, and in old-growth forests. He found that numbers and biomass of predators in the total stream, particularly cutthroat trout, were highest in recent clearcuts, lowest in second growth, and intermediate in old growth. Aho (1976), studying cutthroat trout, and

Lyford and Gregory (1975), studying algae and insects, found similar contrasts in populations between clearcut and old-growth forested sections of the same stream but did not examine stream systems in second-growth forests. Riparian habitats are also critical for mammals and birds (Thomas et al. 1979b).

In summary, old-growth forests dominate both the composition and the function of associated streams. Terrestrial litter is the primary source of energy and nutrients. Woody debris also functions as a major site for fixation of N and as habitat for a broad array of organisms. Logs are also the structural key to the physical and biological stability of a stream. In undisturbed forest, streams are highly retentive of organic materials and nutrients; little escapes without being at least partially processed (consumed and decomposed). Exported material provides downstream reaches with prepared food resources.

The large diameters and heights of the old-growth trees are the most striking structural attributes of these forests. Heterogeneity in diameter and spacing contributes, however, to the variety in an old-growth forest. The pioneers—predominantly Douglas-fir—continue to enlarge in diameter and height over time, while natural thinning reduces their numbers (table 6 and fig. 6). Shade-tolerant species, such as western hemlock, invade the stand and provide smaller trees.

Table 6—Density of all trees and Douglas-fir, mean d.b.h. of Douglas-fir, and stand basal area in age sequence of old-growth Douglas-fir–western hemlock stands in the Cascade Range

Forest	Stands sampled	Density[1] All trees		Density[1] Douglas-fir		Mean d.b.h., Douglas-fir		Stand basal area	
Years	Number	Number per acre	Number per hectare	Number per acre	Number per hectare	Inches	Centimeters	Square feet per acre	Square meters per hectare
250	3	156	389	50	124	32	81	444	83
250[2]	9	NA	NA	52	130	NA	NA	NA	NA
430[2]	7	NA	NA	31	78	NA	NA	NA	NA
450	7	163	407	24	60	47	119	444	102
850	3	222	556	3	8	71	180	362	83

NA = not available.
[1] Trees>2-inch (5-cm) d.b.h.
[2] From Boyce and Bruce Wagg (1953).

Tree density in age series of stands

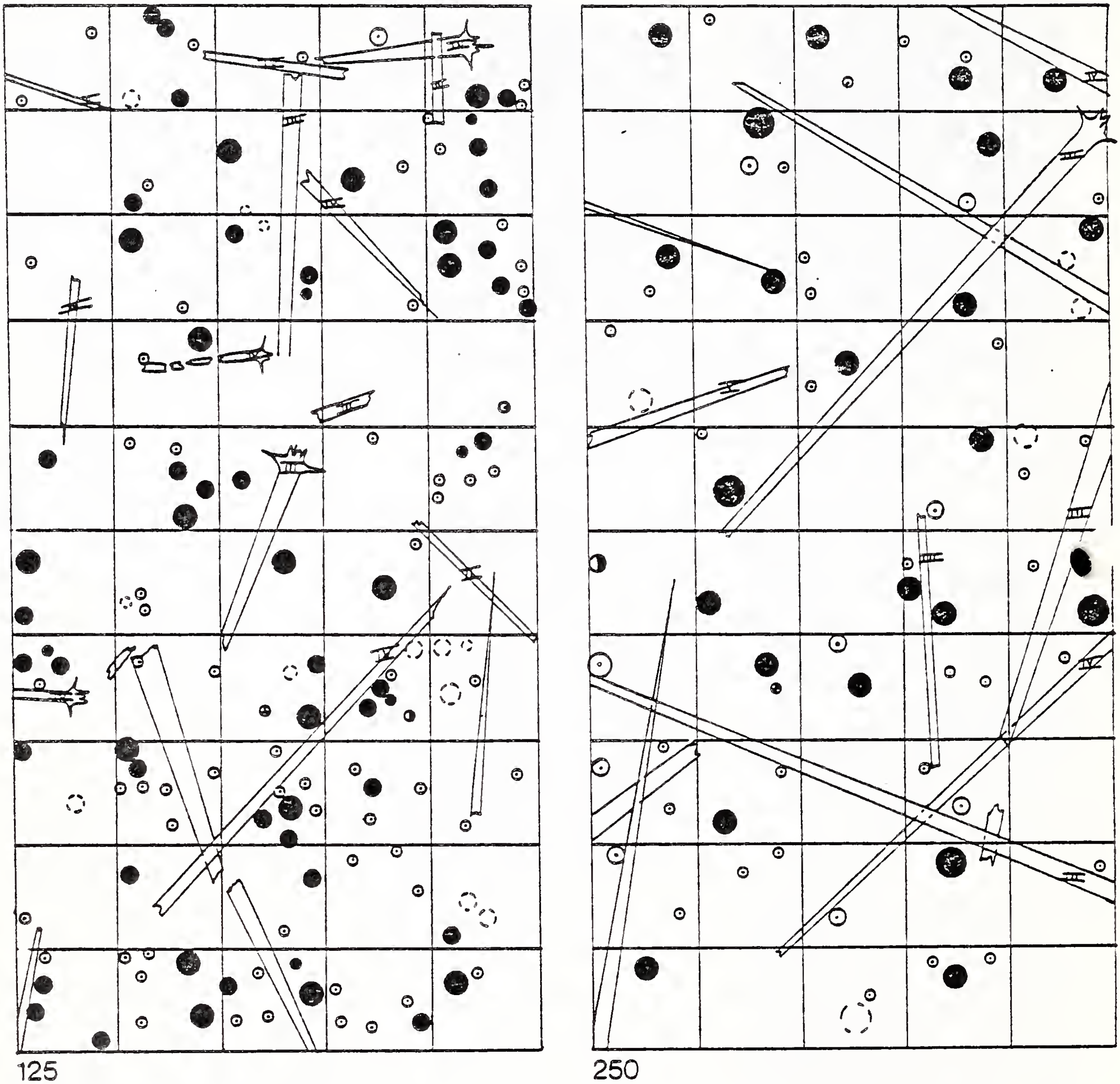
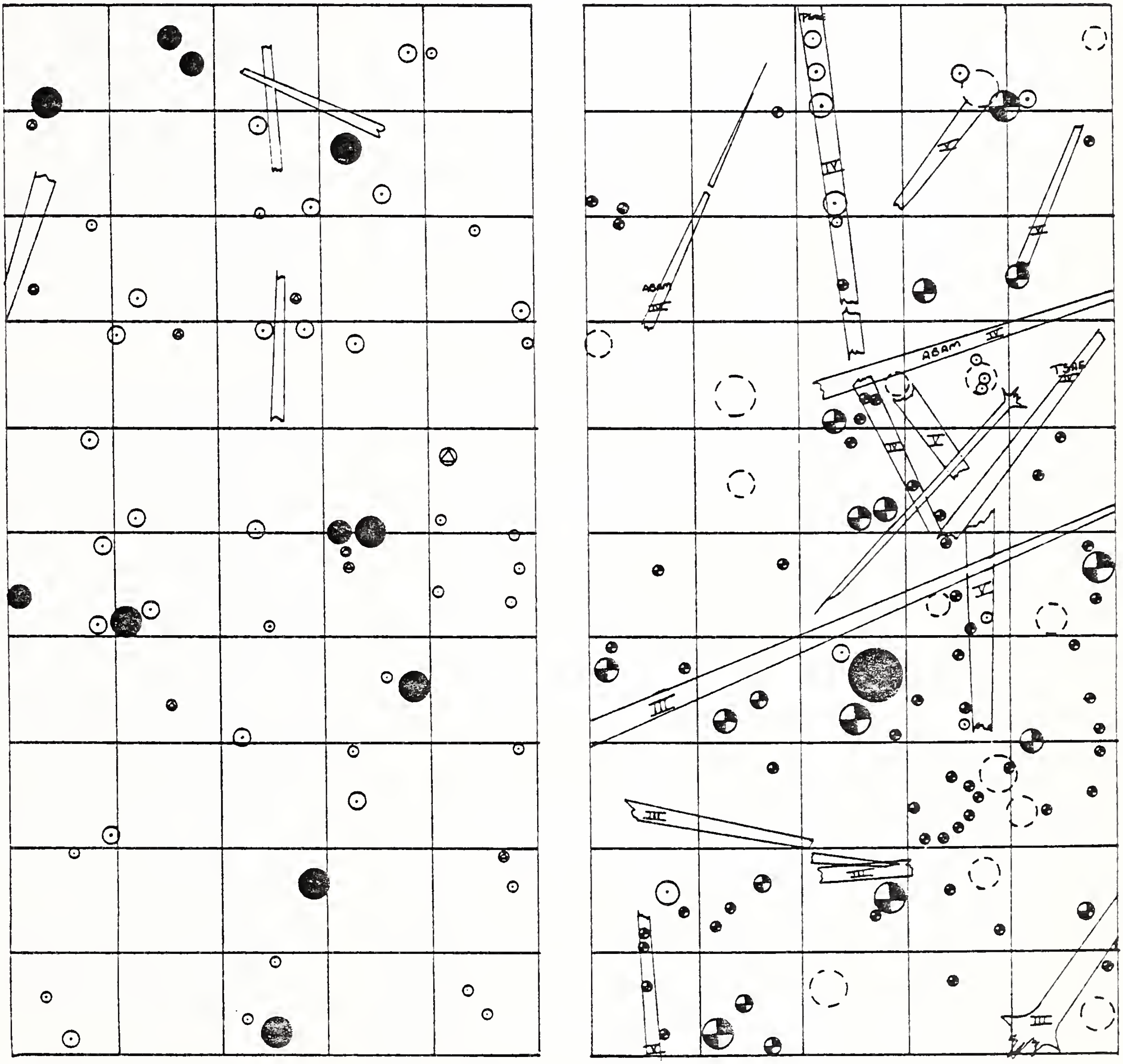


Figure 6. Stem maps of forest stands approximately 125, 250, 450, and 1,000 years of age, suggesting the increased range of tree sizes, reduced densities of Douglas-fir, and shift to shade-tolerant tree associates related to aging of the forest (all live trees greater than 2-inch (5-cm) d.b.h.).

Tree density in age series of stands



450 1,000

Species code:

- *Pseudotsuga menziesii*
- *Tsuga heterophylla*
- *Abies amabilis*
- *Thuja plicata*
- *Taxus brevifolia*

Stem size class (centimeters):

- 0-25
- 26-50
- 51-100
- 100 +
- Standing dead
- Logs

Table 7—Mean diameters and coefficients of variation for all trees in excess of 12-centimeter (5-inch) d.b.h. for 7 young-growth stands and 7 old-growth stands on a wide variety of sites in the central Oregon Cascade Range

	Young growth				Old growth		
	Mean diameter		Coefficient of variation		Mean diameter		Coefficient of variation
	Inches	Centimeters	Percent		Inches	Centimeters	Percent
	11.5	28.7	77		10.8	27.0	131
	12.5	31.3	80		12.3	30.8	112
	13.3	33.3	84		14.4	36.1	100
	13.6	34.0	69		15.8	39.5	91
	16.9	42.2	48		17.4	43.5	114
	20.8	51.9	53		21.7	54.2	92
	25.7	64.2	39		29.2	73.1	61
Overall mean	16.3	40.8	64	Overall mean	17.4	43.5	100

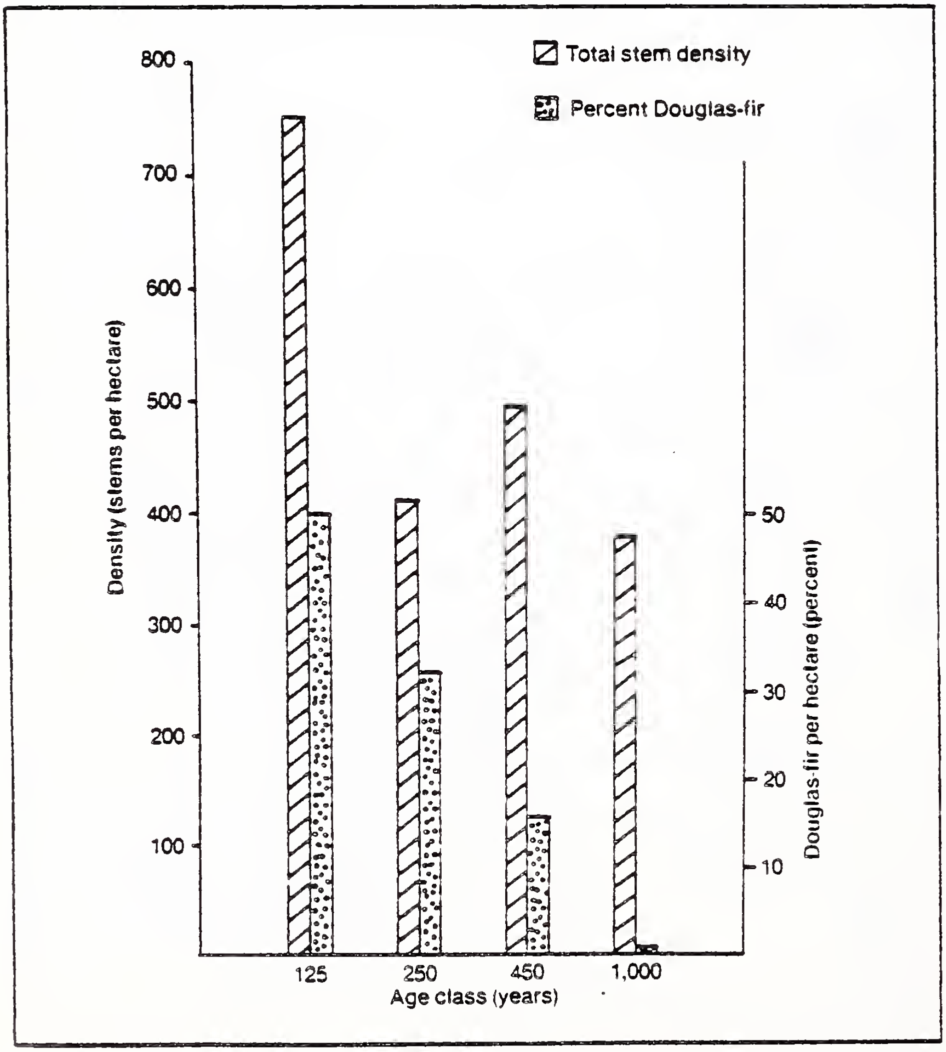


Figure 7. Densities of all trees and Douglas-fir for stands illustrated in figure 6, showing the relatively constant level of density of all trees greater than 2-inch (5-cm) d.b.h. and declining densities of Douglas-fir.

Comparison of old-growth stands (450-year) with young-growth stands (125-year) in the Cascade Range of Oregon shows mean diameter of all trees over 12-inch (5-cm) d.b.h. to be close (table 7). The range of diameters is much greater in the old-growth stands, however, and this is reflected in the larger coefficients of variation (table 7). Total tree density (number of stems >2-inch d.b.h./acre or 5-cm/ha) does not appear to change much after a stand reaches 250 years of age (table 6 and fig. 7). Texture of bark changes with age and size, which, with the associated variation in diameter, undoubtedly reinforces the impression of greater heterogeneity in old growth.

There are few data to support the intuitive prediction that spatial heterogeneity of old-growth forests is greater than that of young growth. Moreover, variation in original stocking levels and site productivity from site to site would obscure any pattern in spacing. We tried to reduce some of this variation by pairing three young-growth and old-growth stands on the same habitat types in the Cascade Range. An analysis of distance to nearest tree was performed on the paired stands and on old-growth stands in the Coast Ranges of Oregon. Mean distance to the nearest tree and coefficients of variation in stands in the Cascade Range are somewhat greater in old-growth forests than in young growth (table 8). Because of unequal size of samples and their small numbers, the differences are not statistically significant at even the 10-percent level. The small size of the sample may also be obscuring trends. Mean spacing is significantly greater in stands of the Coast Ranges than in the Cascade Range (5-percent level), however, and the coefficients of variation tend to be larger (table 8).

Another way to examine diversity is to plot histograms of numbers of trees in distance to nearest tree classes for the stands (fig. 8). Distributions are skewed much more toward smaller distances to nearest tree in the young-growth stands than in the old growth.

A commonly used index of diversity (H' or the Shannon-Weaver index) was calculated for the stands shown in figure 8 from the histogram data. Higher values of this index indicate greater diversity—the young-growth stands have H' values of 2.037 and 2.212; old-growth stands, 2.332 and 2.443.

Conclusive demonstration of trends toward increased spatial heterogeneity in old-growth forests will require a much larger number of samples along a chronosequence of Douglas-fir stands, as well as in other coniferous forest types.

Diversification of tree structure may begin early. Many 90- to 130-year-old stands begin to show greater ranges in size of trees and a multilayered canopy. Time for development can also vary substantially with site conditions; stands on moist, productive sites develop a wider range of sizes earlier than do stands on dry, less productive sites. The broad range of sizes and varied canopy (as opposed to the monolayer of Douglas-fir canopies in young-growth stands) do not generally become well developed, however, until stands reach 200 to 250 years of age.

Table 8—Mean distance to nearest tree and coefficient of variation for all trees over 2-inch (5-cm) d.b.h. in 3 young-growth and 3 old-growth stands on comparable habitat types in the Oregon Cascade Range and in 7 old-growth stands in the Oregon Coast Ranges

Location	Young growth (125 years) Mean distance		Young growth Coefficient of variation	Old growth (450 years) Mean distance		Old growth Coefficient of variation
	Feet	Meters	Percent	Feet	Meters	Percent
Cascade Range	7.00	2.12	37	9.14	2.77	53
	6.30	1.91	67	8.58	2.60	46
	9.24	2.80	49	9.47	2.87	45
Coast Ranges				9.14	2.77	53
				7.00	2.12	63
				9.11	2.76	55
				7.19	2.18	57
				10.56	3.20	65
				10.43	3.16	59
				10.00	3.03	58

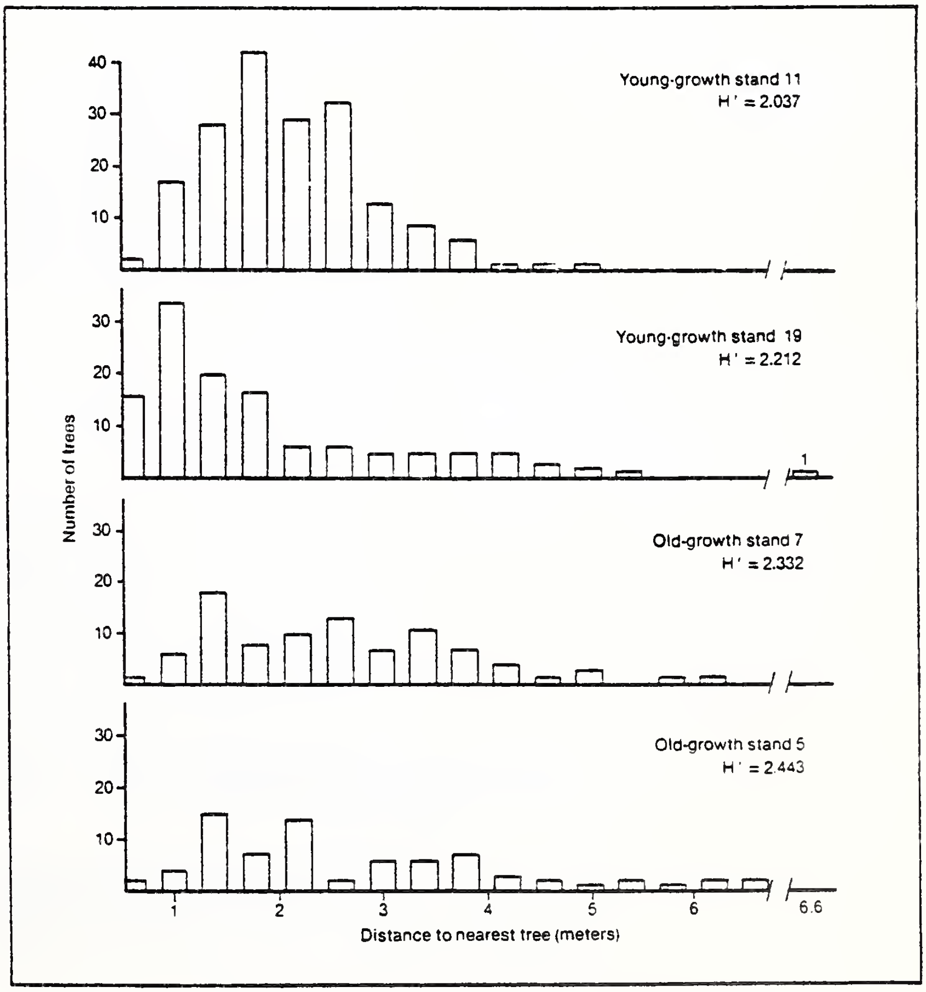


Figure 8. Number of trees greater than 12-inch (5-cm) d.b.h. by distance to nearest tree classes in two young-growth (125 years) and two old-growth (450 years) stands in the Cascade Range of Oregon.

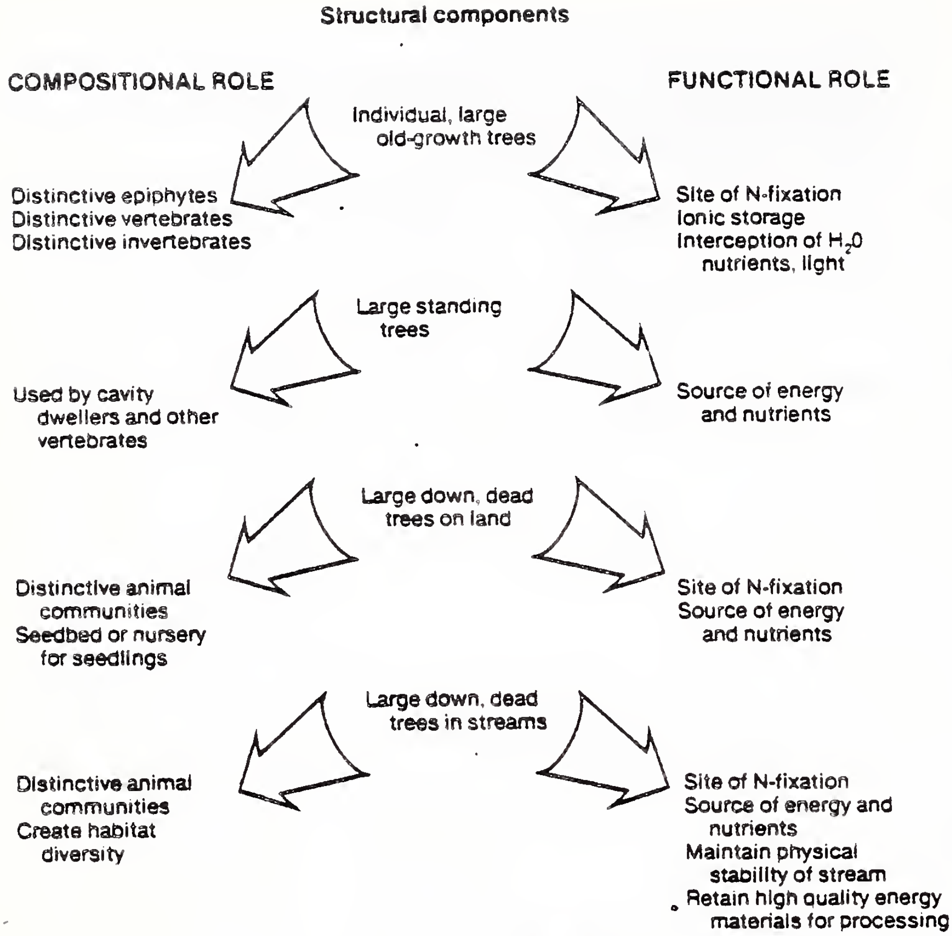


Figure 9. Most of the special habitat and ecosystem functions of old-growth conifer forests can be related to four structural components.

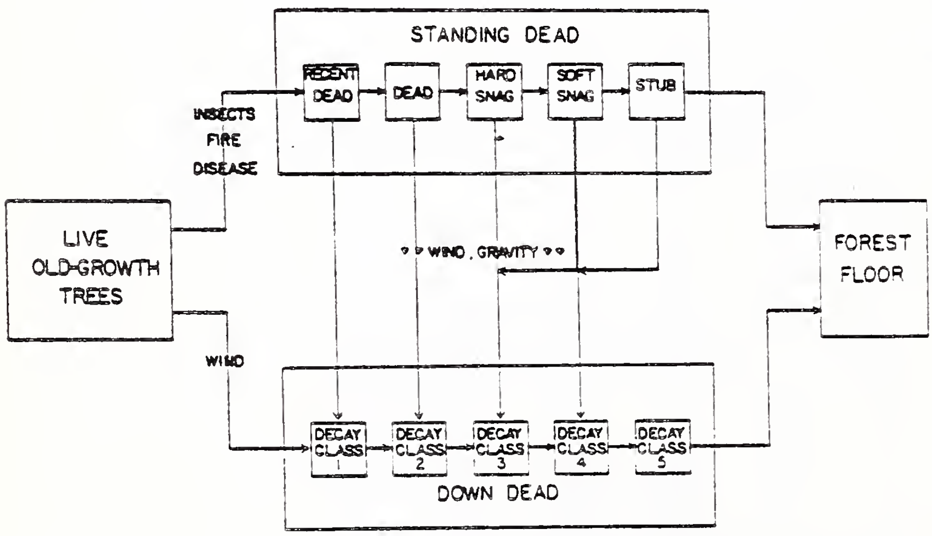
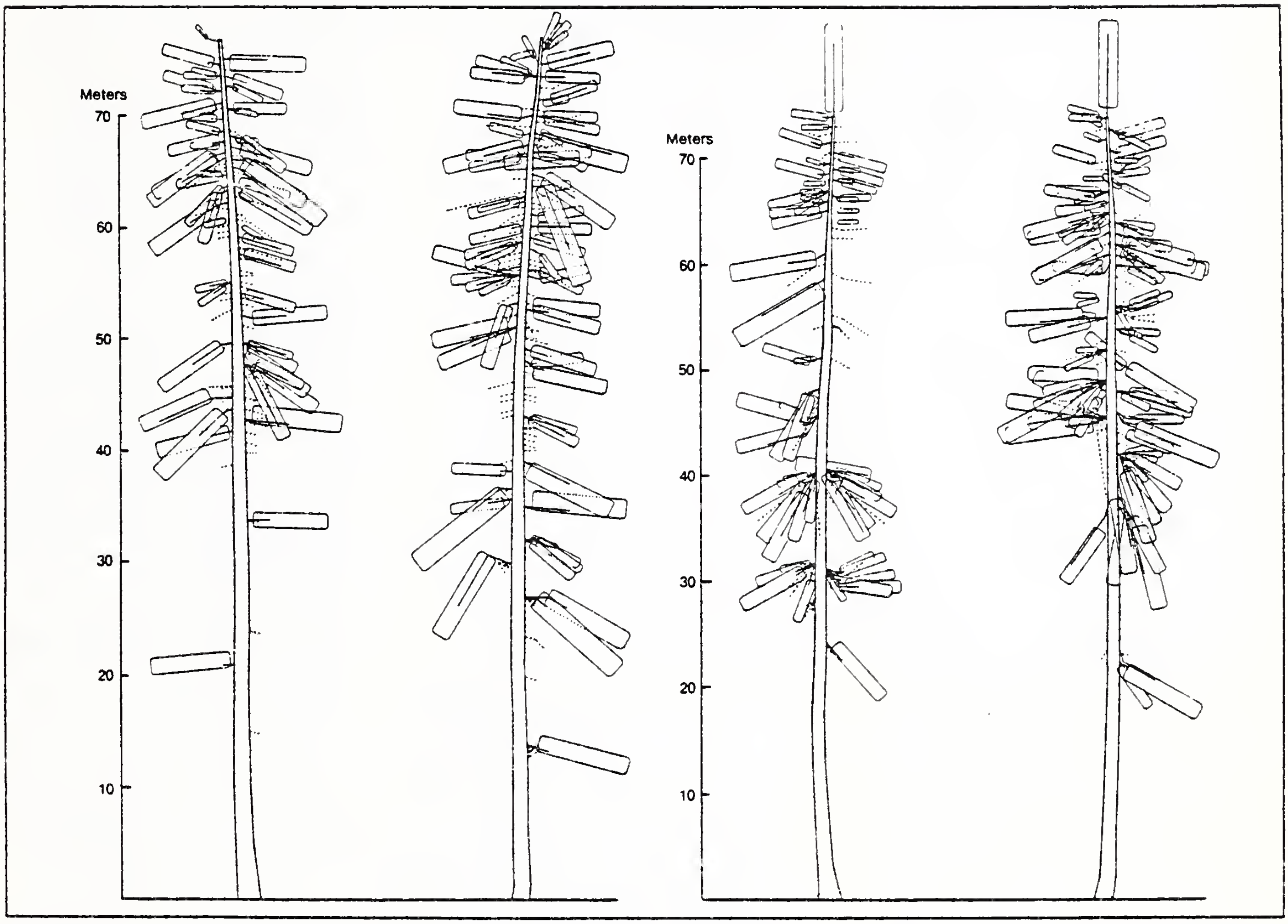


Figure 10. Routing of tree stems from live through various dead organic compartments; many pathways are possible.

A phenomenon related to diversification of stand structure is the development of greater patchiness in understory tree seedlings, shrubs, and herbs. Many young stands have relatively uniform understories—whether extremely depauperate, as in very dense stands, or with continuous cover of some dominant understory species, such as salal or swordfern. Homogeneity in the understory vegetation and forest floor gradually disappears as a stand develops. Many factors are responsible—shifting patterns of openings and heavily shaded areas and provision of new substrate, such as windthrown tree trunks and root wads. Understory patchiness seems characteristic of old growth in other forest types (coastal and subalpine), as well as Douglas-fir stands.

Three structural components (or four, counting logs on both land and in streams) are of overwhelming importance in an old-growth forest. These are the individual, live, old-growth trees; the large, standing, dead trees or snags; and the large, dead, down trunks or logs. Logs are at least as important (and possibly more so) to the stream component of the ecosystem as they are to the terrestrial component. It is these structural features that are, in large measure, unique to an old-growth forest ecosystem, setting it apart from young growth and, especially, managed stands. Furthermore, most of the unique, or at least distinctive, compositional and functional features of old-growth forests can be related to these structural features (fig. 9); that is, these structural components make possible much of the uniqueness of the old-growth forest in terms of flora and fauna (composition) and the way in which energy and nutrients are cycled (function).



It is important to recognize that the four structural components are related (fig. 10). The dead organic structures—large snags and logs—are derived from the live old-growth trees. The tree thus plays a progression of roles from the time it is alive through its transformation to an unrecognizable component of the forest floor.

Live Old-Growth Trees. The most conspicuous of the four key structural components is probably the live, old-growth Douglas-fir trees. These trees are large; though size varies with site conditions and age, diameters of 3 to 6 feet (1 to 2 m) and heights of 165 to 295 feet (50 to 90 m) are typical. They are highly individualistic, much less uniform than trees in a 50- to 150-year-old stand. Each has been shaped over the centuries by its genetic heritage, site conditions, competition with nearby trees, and the effects of storms, diseases, insects, and, possibly, soil mass movement.

The large, deep, irregular crown, characteristic of many old-growth Douglas-fir trees, is as ecologically important as the massive trunk (fig. 11). A 450-year-old tree typically has the overall shape of a bottle brush (albeit one with many missing bristles), with a cylindrical crown beginning 65 to 130 feet (20 to 40 m) above the ground and composed of

Figure 11. Schematic profiles of two old-growth Douglas-fir trees illustrating their individualistic nature and deep asymmetric crowns; several fan-shaped branch clusters are identifiable on the lower bole of both trees (diagrams courtesy W. Denison).

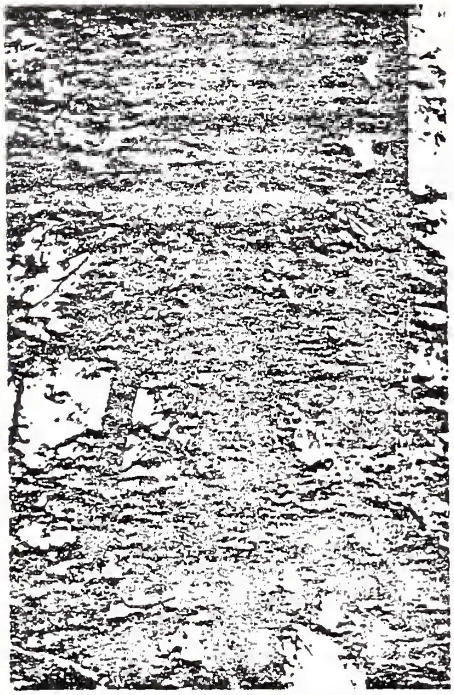


Figure 12. Many lower branches on old-growth Douglas-firs are large, horizontally flattened, fan-shaped arrays arising from the stub of an older branch; these provide extensive horizontal surfaces (photo courtesy W. Denison).

slender branches up to 6-inch (15-cm) diameter.[7] Branches are irregularly scattered through the lower two-thirds of the canopy; often there are gaps of many meters on one side of a tree (fig. 11). Many lower branches are horizontally flattened, fan-shaped arrays arising from the stub of an older branch and showing evidence of repeated breakage (fig. 12). Such massive irregular branch systems may be on one side of the trunk, but their folliced parts can spread out to surround over three-quarters of the circumference of the trunk. Upper surfaces of large branches are covered by organic "soil" (several centimeters thick), which is perched on the branches and supports entire communities of epiphytic plants (mainly mosses and lichens) and animals. Large branches are the home of myriad invertebrates, as well as birds and arboreal mammals. Branches in the upper third of the canopy are more numerous and regular in shape; they resemble those of younger trees.

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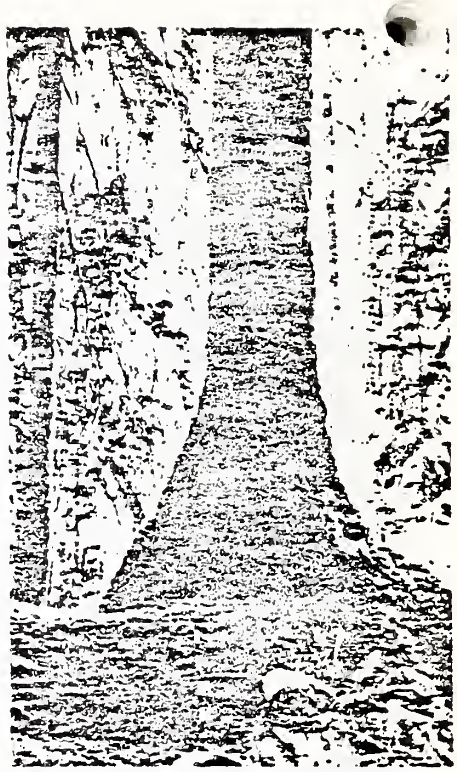
[7] The deep crowns of many old-growth Douglas-firs in old-growth forests of the Cascade Range have been the subject of considerable discussion among the authors. Douglas-fir crowns in many natural 75- to 150-year-old stands are quite short; live branch systems are confined to the upper one-third to one-fourth of the bole. It is hard to imagine how these trees could develop crowns similar to those of existing old-growth trees, even after several centuries. Epicormic branching is probably one factor. We also think that existing old growth originally developed in stands that were understocked; under such conditions, branch systems might persist much lower in the crown. A wide range in ages of dominant old-growth Douglas-firs in many stands does provide some evidence of low densities in original stands (Franklin and Waring 1980). If our inference about low tree densities being a factor in old-growth crown forms is correct, one implication would be that dominant Douglas-firs in many existing second-growth stands would not develop the "classical" old-growth form of crown. They might, however, develop the alternate form of crown often observed in the Coast Ranges and discussed later in this report.

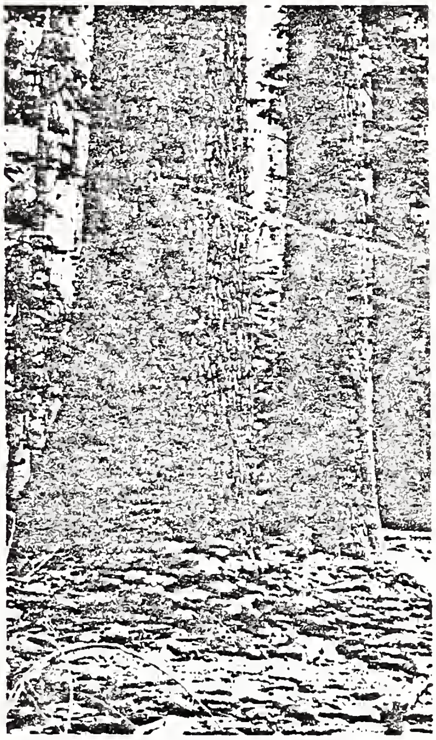


A

Figure 13. Long-lived species in the other forest types can attain the sizes of old-growth Douglas-fir and fulfill comp[illegible] ecological roles: Old-growth Sitka s[illegible] (A) in coastal forests attain diameters of 60 inches (150 cm) or more and have massive epiphyte-laden branch systems. Noble fir (B) attains the largest size of any subalpine species but lacks a deep crown and dense branch system. Old-growth western redcedar (C) and Port-Orford-cedar (D) can be large specimens functionally comparable to old-growth Douglas-fir in many respects.

Old-growth trees frequently have broken tops, in which case one or several lateral branches may have grown upward and assumed leadership. Lateral branches of these secondary tops resemble those in the upper portion of an intact top.

Most stands have occasional old trees with their crowns concentrated at the top of the trunk in a spherical rather than a cylindrical form. Such



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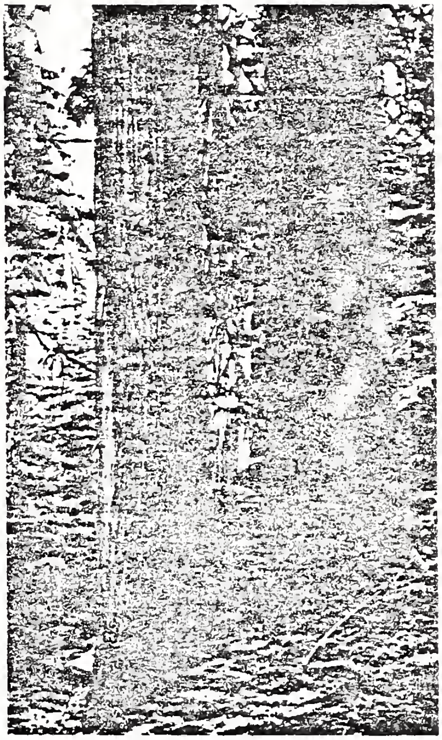

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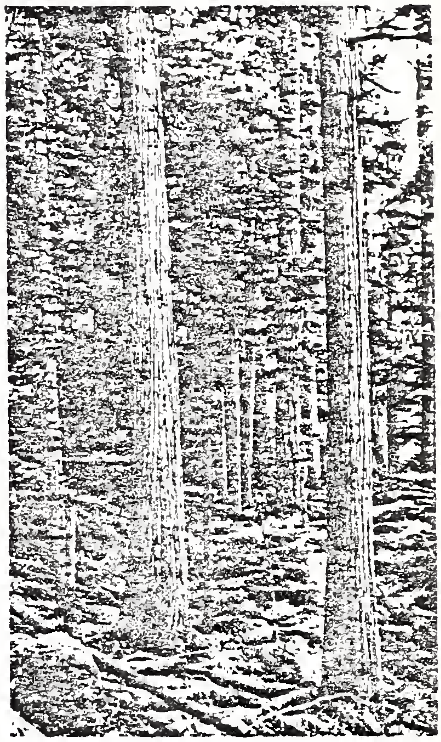


D

trees often overtop the adjacent canopy. The crowns of these trees are dominated by much larger limbs than are found in cylindrical crowns. This spherical form of crown seems more characteristic of old-growth stands in the Coast Ranges than of those on the west side of the Cascade Range and may reflect crowns developed in a denser stand (see footnote 7).

Few old-growth Douglas-firs have vertical trunks. The lower trunk leans away from the hillside but becomes nearly vertical where it extends above the surrounding canopy. Trunks on level sites appear to slope almost at random. Even a slight inclination of a trunk results in an important differentiation of habitat on its two sides. The upper side gets almost all the moisture, both from direct precipitation and from stem flow and throughfall. Consequently, it is colonized by epiphytic plants (plants that grow on other plants) with relatively high moisture requirements, chiefly mosses. The lower side is a "desert" occupied by scattered colonies of lichens (Pike et al. 1975) that form a crust over the bark surface. The bark on the wet upper side is soft and easily eroded, sometimes appearing to be held in place by its mantle of mosses and lichens, whereas the bark on the lower side is hard and deeply furrowed, indicating that it remains in place for longer periods.

Old specimens of other tree species can play a role comparable to that of Douglas-fir to at least some degree, although none have been as thoroughly studied (fig. 13). Sitka spruce attains comparable sizes in coastal regions; irregular crown systems and heavy, epiphyte-laden branch systems are characteristic of older specimens. Noble fir and western white pine are subalpine species with some, but not all, of the distinctive characteristics of Douglas-fir, as is sugar pine in southwestern Oregon. The so-called cedars—western redcedar, Alaska-cedar, Port-Orford-cedar, and incense-cedar—are capable of attaining sizes and fulfilling roles comparable to Douglas-fir in their respective types. These species have the additional advantage of fostering improved soil conditions through their base-rich litter. The major climax species—western hemlock, Pacific silver fir, and grand fir—appear, on the other hand, to lack the ability to completely fulfill the ecological roles of these long-lived pioneers.

Table 9—Temperature regime in an old-growth Douglas-fir tree canopy as related to precipitation during the current and preceding day[1]

Precipitation, current and preceding day	Days		Maximum daily temperature		Daily temperature range	
	Sept. 1976 to Aug. 1977	Sept. 1977 to Aug. 1978	Mean	Standard error	Mean	Standard error
Millimeters	Number		°Celsius			
0	195	124	18.8	± 11.2	13.1	± 6.8
0.1- 10.0	91	111	11.7	± 7.8	7.8	± 4.8
10.1- 20.0	35	48	9.5	± 6.0	5.8	± 4.0
20.1- 30.0	18	34	8.8	± 4.8	5.3	± 3.3
30.1-100	26	48	7.5	± 3.7	4.0	± 1.9

[1] An increasingly wet canopy results in lower minimum temperatures and a smaller diurnal range in temperature.

What ecological roles does a live, old-growth tree fulfill? There are many, but the most important are related to: (1) provision of habitat for the distinctive epiphytic plant and animal communities found in an old-growth forest; (2) effects on carbon, nutrient, and water cycling, especially the N budget of the site; and (3) source material for the other key structural components—standing dead trees, logs on land, and logs in streams.

Habitat Function. Many of the distinctive compositional features of old-growth forests—plants and animals—are related to the tree canopies. Almost every surface of an old-growth Douglas-fir is occupied by epiphytic plants; more than 100 species of mosses and lichens function as these epiphytes. The dry weight of mosses and lichens on a single old-growth tree ranges from 33 to 66 pounds (15 to 30 kg) (Pike et al. 1977), of which less than half is mosses; this excludes the ubiquitous crust-forming lichens which cannot be separated for weighing. In forests below 3,500-foot (1 000-m) elevation, about half the total weight of epiphytes is usually due to a single leafy or "foliose" lichen, *Lobaria oregana*, which is an active N fixer. Although lichens are found over almost all surfaces, many species are restricted to particular habitats (see table 1 in Pike et al. (1975) for an excellent illustration of this point). *Lobaria oregana*, for example, occurs chiefly on the upper sides of branches and twigs. *Lepraria membranacea*, on the other hand, prefers the lower trunk and the underside of branches. Nearly all mosses occur on the bottom half of a tree.

Epiphytic communities remove soluble mineral nutrients from water flowing over them. They also trap dust and litter fragments, including needles. This accumulation, augmented by decomposition of the epiphytes themselves, is most evident on the upper sides of large branches where it results in the formation of perched "soils."

When moist, the old-growth forest canopy is an important climatic buffer, a fact that may explain some of the special compositional and functional features of the canopy. Air temperatures in the canopy of an old-growth Douglas-fir stand in the western Cascade Range of Oregon can range as high as 104 °F (40 °C) during the summer and as low as 14 °F (– 10 °C) during dry periods in the winter. When the canopy is wet, however, temperatures range from 32 °F to 60 °F (0 °C to 15 °C) (table 9). As precipitation increases, daily maximum temperatures and the daily temperature range decrease. This buffering reflects the large water-holding capacity of the canopy—about 264,000 gallons per acre (3×10^6 liters/ha)—equivalent to 1¼ inches (3 cm) of precipitation.

This environmental regime is important to survival of *Lobaria oregana* and may be to other canopy inhabitants. *Lobaria*, the dominant epiphytic lichen in old-growth stands on the west slope of the Cascade Range, is metabolically active when wet and dormant when dry. One-half to 1 inch (1 to 2 cm) of rainfall will wet the canopy sufficiently to raise the water content of the *Lobaria* above 70 percent. Below this moisture level, the lichen ceases to

fix N and is presumably protected against temperature extremes by dormancy. A moistened thallus would never be subjected to high temperatures because of the canopy's buffering (fig. 14). *Lobaria oregana* appears to be limited to habitats where moist conditions are always associated with cool temperatures, such as is characteristic of an old-growth canopy. When *Lobaria* thalli are transplanted to stands of young growth or mixed conifer-hardwood, they deteriorate rapidly, presumably because air temperatures exceed 60 °F (15 °C) and thalli are hydrated. *Lobaria oregana* usually does not occur in young Douglas-fir stands, possibly because their canopies hold insufficient moisture for adequate thermal buffering. It may be abundant on individual young trees in old-growth stands, however, where the surrounding mature trees provide an appropriate microclimate.

The canopy of an old-growth Douglas-fir forest harbors large numbers of invertebrates of many species. A single stand may have more than 1,500 species. A minority of species spend their entire cycle in the canopy: Araneida, Acarina, Homoptera, Collembola, Neuroptera, Thysanoptera, and Psocoptera. Other species of Lepidoptera, Hymenoptera, Diptera, and Coleoptera occur as eggs, larvae, and pupae in the canopy; but the adults can and do move out of the canopy. The majority of species encountered in the canopy are adults that spend their immature stages on the forest floor or in streams. In their canopy studies, Drs. George Carroll (University of Oregon) and William Denison (Oregon State University) discovered overwintering caddisfly adults in Douglas-fir canopies. Many adults of species of Mycetophilidae (fungus gnats) trapped in the canopy occur as larvae in the abundant mushrooms on the forest floor.

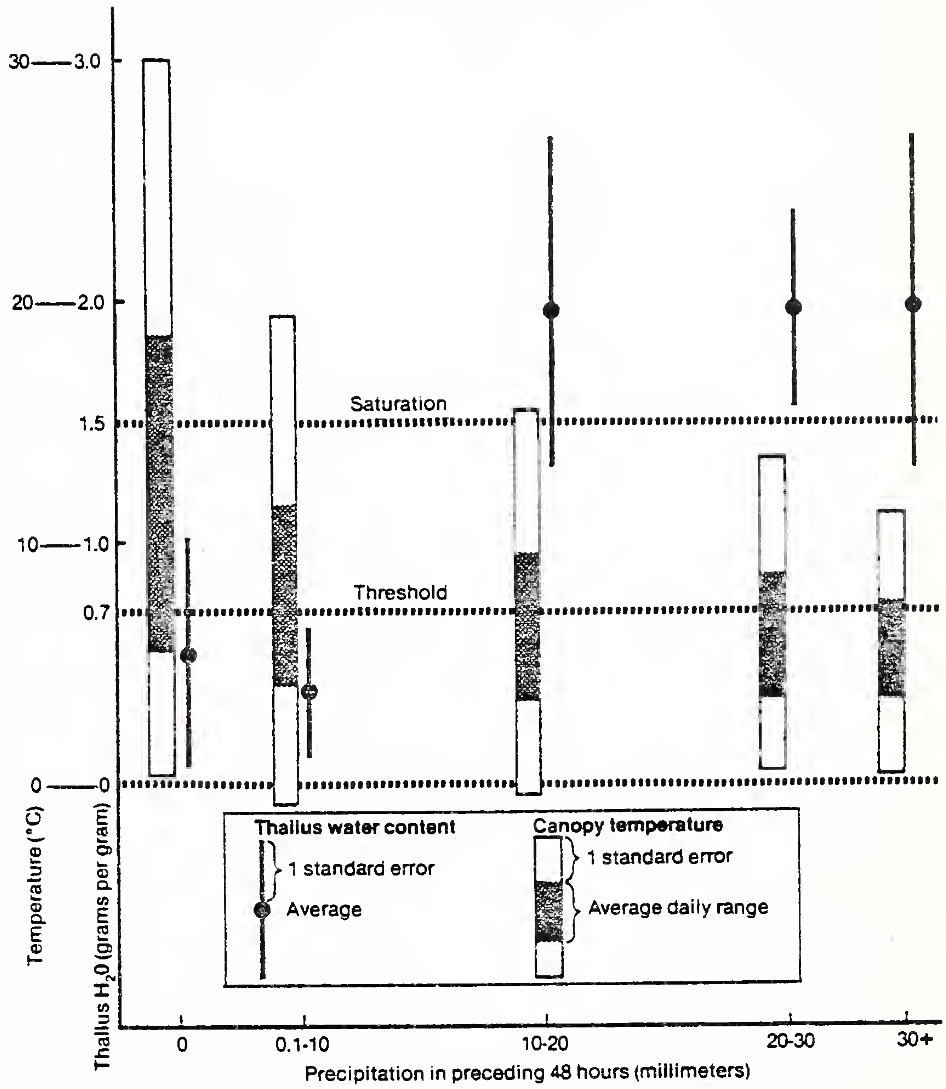


*Figure 14. Relationship between canopy temperature, lichen (*Lobaria oregana*) thallus water content, and precipitation in the preceding 48 hours. Several thresholds are indicated: 0°C which is the lower thermal limit for nitrogen fixation; 70-percent thallus water content which is the lower moisture limit for nitrogen fixation; and 16°C which is the upper thermal limit (tolerance) for a saturated* Lobaria *thallus.*

Although primary consumers (insects—such as sawflies, scales, or aphids—which feed on foliage or beetles which feed on wood) do occur in the canopy, they are not abundant. The most abundant arthropods are predaceous spiders, which belong to families such as Salticidae and Thomisidae. The large numbers of flies found in the canopy probably provide food for the spiders. Other arthropods feed on debris or on bacteria and fungi on surfaces of the canopy or are predators of other invertebrates. During one sampling period, invertebrates washed from foliage samples included:

Food source	*Invertebrates*
Needles	1 species of scale 1 species of mealy bug 1 species of Lepidoptera
Bacteria and fungi living on needles	6 species of mites 4 species of flies (as larvae) 5 species of Collembola
Other invertebrates	2 species of mites 2 species of spiders

The canopy of an old-growth forest provides several insect habitats, both vertically and horizontally. Some species are found in the upper canopy, others in the lower; some species occur on major limbs and others among twigs and foliage.

Several vertebrates depend heavily on the old-growth canopy as sites for nesting, feeding, and protection. Well-known examples are the northern spotted owl, northern flying squirrel, and red tree vole. The vole may live for many generations in the same tree. The role that the large branch systems and organic accumulations play in providing suitable habitat should not be overlooked.

Cycling Function. Old-growth trees are one of the primary sites for photosynthesis, or production of the food base, on which the rest of the system depends. In this sense, they are the same as younger trees, except that each tree represents a large accumulation of organic material and nutrients (a "sink" in the short run and a "storehouse" in the long run) as well as a large photosynthetic factory. A single old-growth tree can have over 60 million individual needles with a cumulative weight of 440 pounds (200 kg) and a surface area of 30,000 square feet (2 800 m^2) (Pike et al. 1977). Total leaf areas in old-growth stands are probably not much different from those in younger stands, but the leaves are concentrated on fewer individuals. Fluctuations in production and live biomass strongly reflect mortality of these large dominant trees, which are both factory and storehouse, and the rate at which other trees occupy the vacated space.

A distinctive and unusual functional role of an old-growth tree is its contribution to the nitrogen economy of low-elevation to midelevation sites. Lichens that inhabit the canopy fix significant amounts of N which ultimately become available to the whole forest through leaching, litter fall, and decomposition. Estimates of fixed N range from 2.5 to 4.5 pounds per acre (3 to 5 kg/ha) per year. Most of the fixation is accomplished by *Lobaria oregana*, although several other large foliose lichens, such as *L. pulmonaria*, *Pseudocyphellaria rainierensis*, and *Peltigera aphthosa*, are also azotodesmic[a] and, therefore, capable of fixing atmospheric N. *Lobaria oregana* accounts for half the total epiphytic biomass in the western Oregon Douglas-fir stands that have been studied. *Lobaria* and most N-fixing epiphytes are not common in young-growth stands, and this may be related to the microclimate of the old-growth forest canopy. Significant epiphytic inputs of N are, therefore, largely confined to old growth. Nitrogen-fixing bacteria on Douglas-fir foliage have not been found in the Pacific Northwest, even though they have been reported in Europe.

[a] Azotodesmic lichens contain a blue-green alga, either as a primary plant symbiont or a secondary one, and therefore are capable of fixing N. Nonazotodesmic lichens contain a green alga as the sole algal symbiont and are not capable of fixing N.

Standing Dead Trees or Snags. In any old-growth stand there are substantial numbers of standing dead trees or snags (fig. 15). Indeed, snags were the first dead component of natural forests of which foresters were made aware—initially because of the fire and safety hazard they represent and, more recently, because of their value to wildlife (Bull and Meslow 1977, Bull 1978, Thomas et al. 1979a, Mannan et al. 1980). Some representative data for old-growth stands are provided in table 10. The only comprehensive study on dynamics of snags is by Cline et al. (1980), who studied 30 stands from 5 to 440 years old in the Coast and Cascade Ranges. Densities of snags decrease with stand age, but mean d.b.h. of snags increases from 5 to 29 inches (13 to 72 cm) between stand ages 35 and 200; larger snags survive longer. Cline et al. (1980) report mean densities of snags over 3.6-inch (9-cm) d.b.h. at 13.8 per acre (34.6/ha) and 7.3 per acre (18.3/ha) in stands 120 and 200 years old, respectively. These values, as well as a life table (model) estimate of 9.2 snags per acre (23/ha) for a 200-year-old stand, are substantially below the densities in table 10; all six of the old-growth stands of Cline et al. (1980) are located in the Coast Ranges.

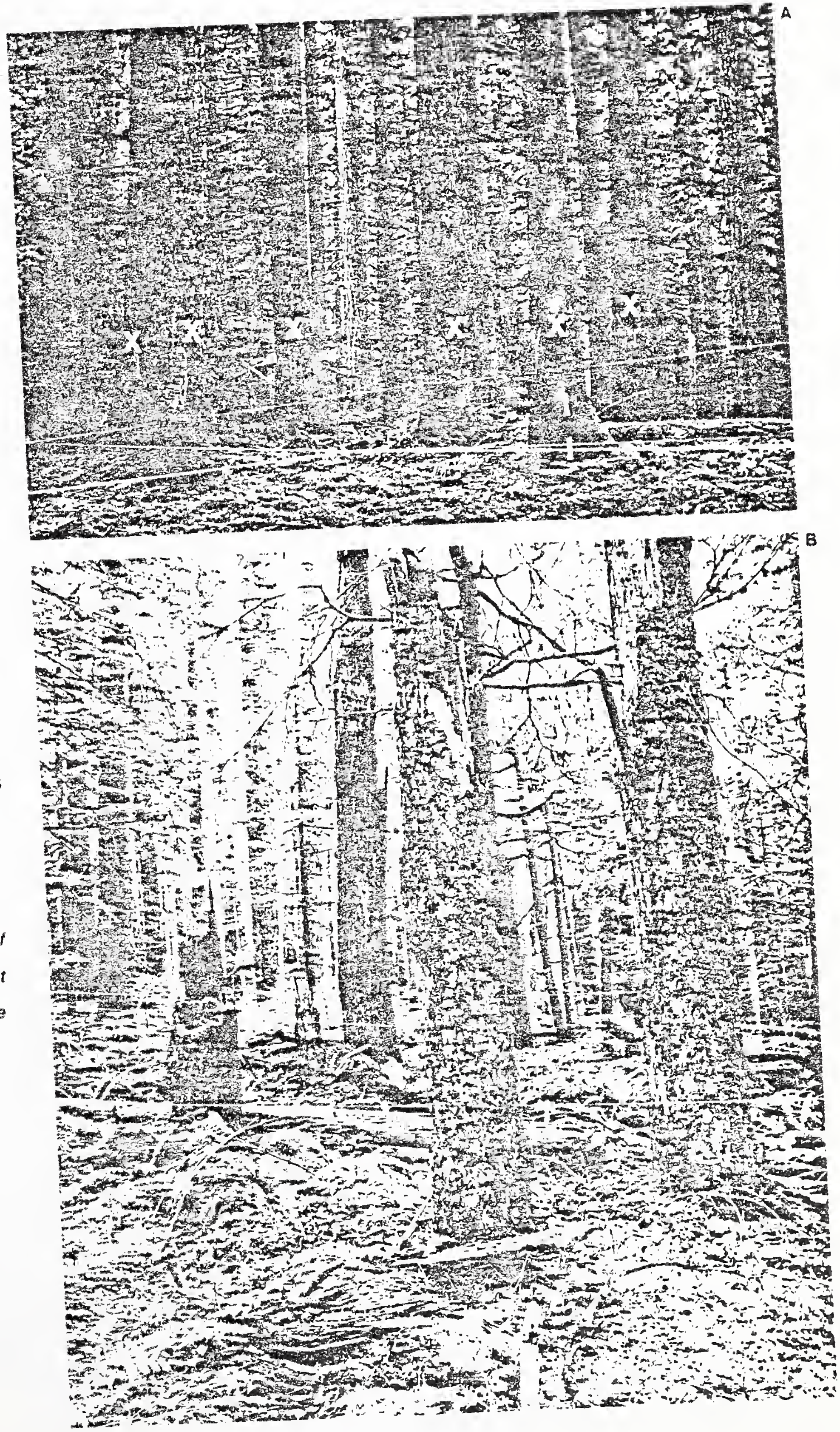


Figure 15. Large numbers of standing dead trees or snags are characteristic of old-growth forests. A. The volume and numbers of standing dead trees may not be apparent to the casual observer in this 250-year-old Douglas-fir stand in the Bagby Research Natural Area, Mount Hood National Forest; dead stems are marked with an X. B. Heavily decomposed snags in old-growth Douglas-fir–western hemlock stand.

Table 10—Numbers of standing dead trees >13 feet (>4 m) in height and mean d.b.h. in age sequence of old-growth Douglas-fir–western hemlock stands in the Cascade Range[1]

Forest age	Stands sampled	Height: 13-31 feet (4-9 m)	Height: 32-64 feet (10-19 m)	Height: >65 feet (>20 m)	Height: All	Mean d.b.h.
Years	Number	Number/acre (number/ha)				Inches (centimeters)
250	2	10 (26)	9 (22)	5 (12)	24 (60)	16 (42)
450	6	6 (16)	4 (10)	2 (6)	13 (32)	22 (57)
850 +	3	7 (17)	5 (12)	2 (4)	13 (33)	25 (64)

[1] Short snags or stubs (<13 ft or <4 m in height) average about 61 per acre (152/ha) in 7 old-growth stands ranging from 250 to 1,000 years old; there is no apparent trend in numbers with age of stand in this small sample.

The large standing dead stems in excess of 20-inch (50 cm) d.b.h. and 65-foot (20-m) height are most valuable to wildlife (Scott 1978). Mannan et al. (1980) found hole-nesting birds usually used snags over 24-inch (60-cm) d.b.h. and 50 feet (15 m) tall in western Oregon. Density and diversity of species of hole-nesting birds were significantly related to mean diameter of snags. Smaller snags apparently do not provide suitable habitat for some animal species, and some tree species are preferred by hole-nesters (McClelland et al. 1979). Under natural conditions, large snags are not strictly a unique attribute of old-growth stands. Young-growth forests developing after wildfires have large residual snags from the original stand for various lengths of time. Cline et al. (1980) found residual or remnant snags in young-growth forest up to the oldest (110-year) age class they studied. Our experience is that large Douglas-fir snags typically persist for 50 to 75 years before being reduced to stubs less than 35 feet (10 m) in height; snags of western redcedar may remain essentially whole and standing for 75 to 125 years.

Large snags result from large trees; so they are a special product of old-growth forests. Managed young stands lack the residual snags of postwildfire stands unless snags are specifically planned. Natural stands appear to require about 150 years to develop snags 20 inches (50 cm) in diameter (Cline et al. 1980). Cline et al. (1980) suggest a life table approach for predicting densities and sizes of snags and recommend retention of large, defective trees for future snags in second-growth forests.

Various classifications, based on external features, have been developed for snags (Cline 1977, Cline et al. 1980, Thomas et al. 1979a); in general, these describe a time sequence in decomposition and disintegration of a dead tree (fig. 16). It is important to differentiate the stages of a snag since these are associated with changing values for wildlife. Both the path (stages) and the rate of disintegration of snags vary widely, however, depending on such factors as tree species, incidence and extent of decay at time of death, and environmental conditions. Douglas-fir snags typically disintegrate from the top down, losing the top and bark first. The trunk finally breaks off in large chunks, leaving a short snag or stub. Western redcedar and western white pine, on the other hand, often form bark-free, gray "buckskin" snags and remain essentially entire until they rot away at ground level and fall.

Habitat Function. A primary role of standing dead trees is the provision of habitat for wildlife. This has been discussed by Thomas et al. (1979a) for the Blue Mountains of Oregon and Washington. Snags in that area are the primary location for cavities that are used by 63 species of vertebrates—39 birds and 24 mammals. Uses include sites for nesting and overwintering, locations for courtship rituals, and food sources.

Thomas et al. (1979a) indicate a direct correlation between numbers of snags and related populations since suitable nesting sites are generally thought to limit populations; Mannan et al. (1980) confirm this for hole-nesting birds in western Oregon. The large, hard snags required by primary excavators, such as the pileated woodpecker (*Dryocopus pileatus*), are especially important. Such snags will be hard to perpetuate in managed stands (because of smaller trees and programs for salvaging wood and reducing fire and safety hazards); yet such snags are also suited to other wildlife species and will produce soft snags through the process of deterioration. Snags representing a variety of decay classes are needed in a stand to meet the differing requirements of vertebrates since not all use the same material. One special attribute of old-growth and large (natural), second-growth stands is that they provide the necessary array of snags with varying levels of decay, whereas young stands on cutover areas do not.

Cycling Function. Most of the functional roles (in energy and nutrient cycling, including sites for microbial nitrogen fixation) of standing dead trees are the same as those of logs and will be considered in the discussion of logs.

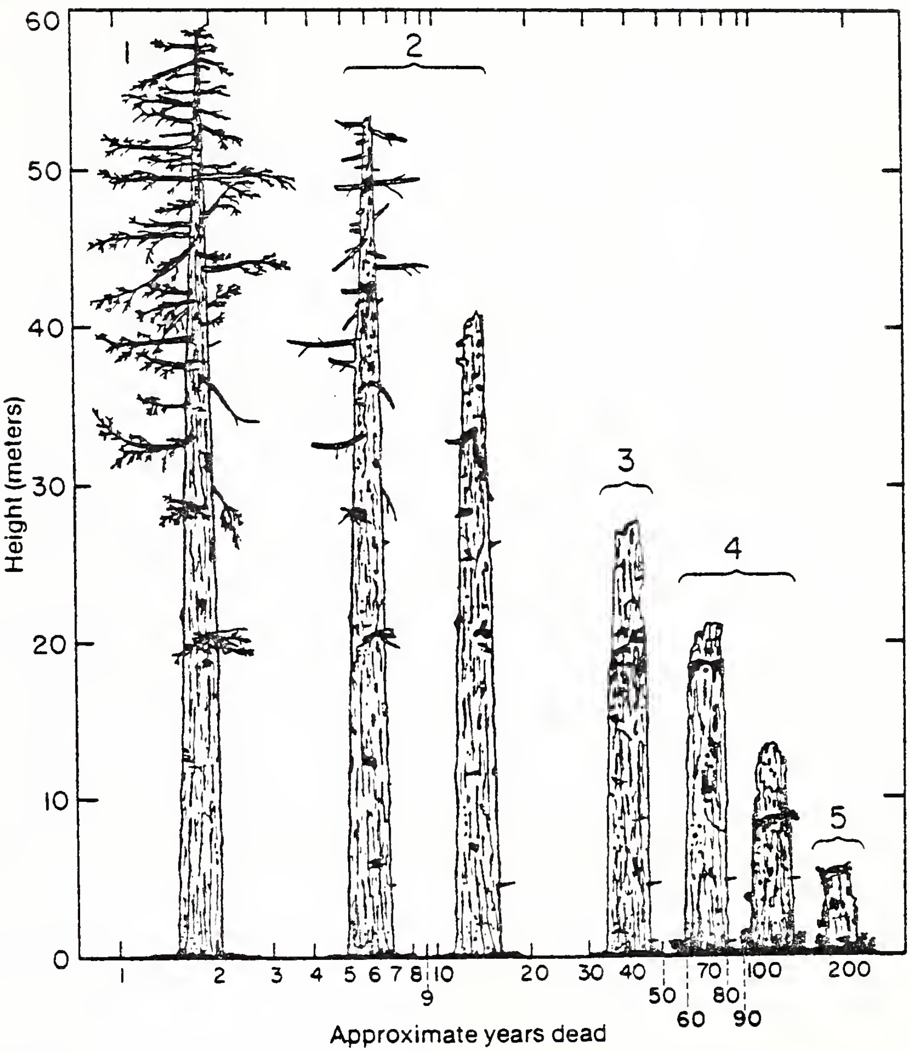
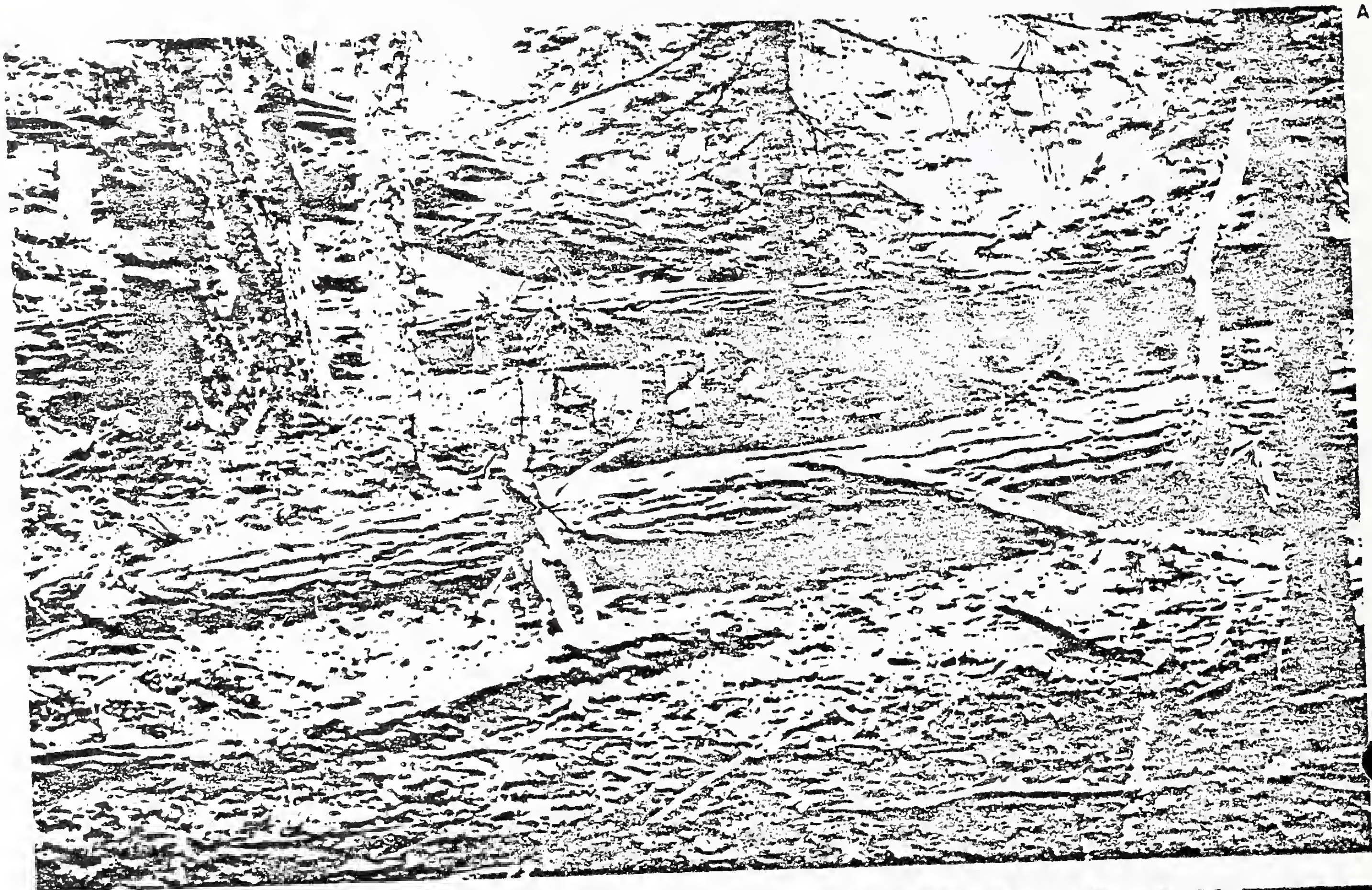


Figure 16. Successional or decompositional evolution of a standing dead Douglas-fir tree (courtesy Steve Cline).

Standing dead trees do not necessarily disintegrate at the same rate or in the same way as logs and cannot be considered simply as vertical, dead trees (or vice versa) in terms of decay rates and agents. An old-growth Douglas-fir that dies standing appears to deteriorate much more rapidly if it remains standing than a tree of comparable size that dies by windthrow. The activity of invertebrate and vertebrate animals, gravity, wind, and the effect of rapidly alternating environmental conditions may all be factors involved in the more rapid disintegration of snags. This difference in the rate and nature of decomposition is, perhaps, the primary functional contrast between down trees and snags.

A



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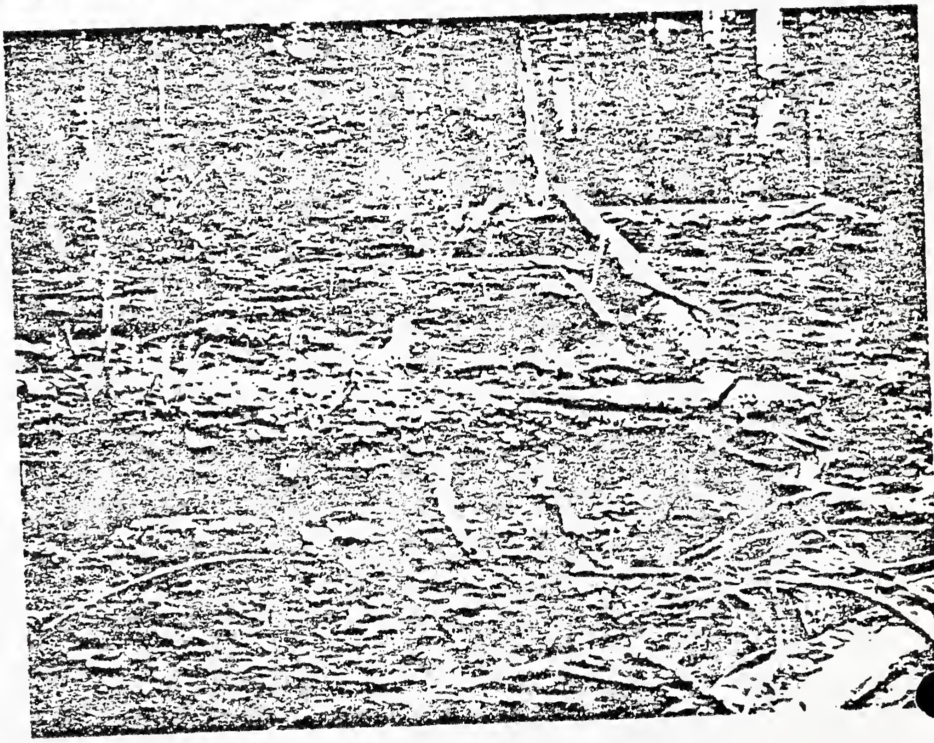

Figure 17. Large masses of logs can be a dominant feature of old-growth forests, as illustrated in these stands with near-maximal accumulations: A. Midelevation stand of old-growth Douglas-fir in the western Cascade Range of Oregon. B. Old-growth stand of noble fir near Mount St. Helens, Washington.

Logs on Land. Logs, also describable as down dead trees or coarse woody debris, are nearly as conspicuous as the large, live trees. Large masses of logs can be the dominant feature of old-growth forests (fig. 17), and, in numbers, volume, and weight of organic matter, they constitute an important component. From 38 to 85 tons per acre (85 to 190 tonnes/ha) are typical values that have been reported. Down logs averaged 85 tons per acre over a 25-acre (10-ha) watershed covered with old-growth Douglas-fir–western hemlock forest (Grier and Logan 1977). Amounts within the watershed ranged widely —the lightest weights (24 tons/acre or 55 tonnes/ha) on a dry ridgetop and the heaviest (259 tons/acre or 581 tonnes/ha) on a lower slope, streamside area. Losses by downslope transfer had occurred on the ridgetop, and substantial amounts of debris had accumulated on the lower slope. In a 10-acre (4-ha) midelevation stand of Douglas-fir, western hemlock, and true firs, there were 82 tons of debris per acre (182 tonnes/ha),[9] 55 percent as recently fallen trees. Logs occupied 29 percent of the forest floor in this stand (fig. 18).

[9] MacMillan, Paul C., Joseph E. Means, Kermit Cromack, Jr., and Glenn M. Hawk. Douglas-fir decomposition, biomass, and nutrient capital in the western Cascades, Oregon. 65 p. Unpublished manuscript on file at Forestry Sciences Laboratory, Corvallis, Oreg.

The average weight of down logs in seven old-growth stands, from 250 to over 900 years old, was 53 tons per acre (118 tonnes/ha);[10] the range was 38 to 70 tons per acre (85 to 156 tonnes/ha).[11] The largest accumulation of down wood recorded for a stand thus far is in the Carbon River Valley at Mount Rainier National Park; a hectare plot contains 188 tons per acre (418 tonnes/ha) of logs that covered 23 percent of the plot.

Logs are also major pools of important nutrients, such as N and phosphorus (P). In the old-growth watershed, the log component contained 192 pounds per acre (215 kg/ha) of N and 6.0 pounds per acre (6.7 kg/ha) of P (Grier and Logan 1977). In the midelevation stand, coarse woody debris contained 485 pounds per acre (544 kg/ha) of N (see footnote 9).

[10] Unpublished data on file at Forestry Sciences Laboratory, Corvallis, Oreg. Stands are at low to middle elevations in the northern Oregon and southern Washington Cascade Range (from H. J. Andrews Experimental Forest, Oregon, to Mount Rainier National Park, Washington).

[11] Weights of down logs and stand age are only loosely correlated. Natural young Douglas-fir stands (about 100 years old), surveyed at the same time as the old growth, had masses of logs as large as some found in old-growth stands—primarily material carried over from previous stands as snags and logs. Large volumes of coarse woody debris are apparently characteristic of our natural forest ecosystems, adding credence to the concept of coarse woody debris as a mechanism to provide continuity of habitat from one forest generation to another (Maser et al. 1979) and for conserving large masses of organic matter and nutrients in major disturbances. It also suggests that the long-term ecological effects of nearly complete removal of woody debris in cutover stands and the prevention of new accumulations in intensively managed stands should be carefully examined.

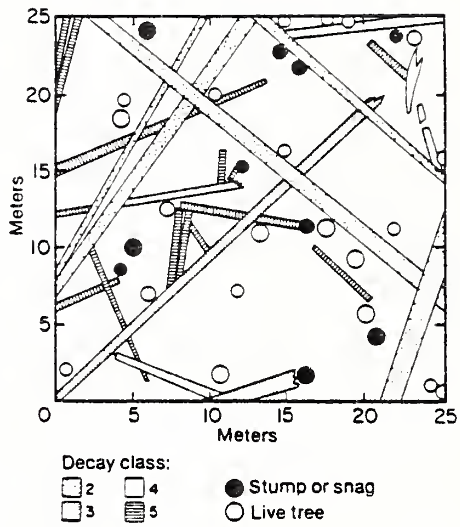

Figure 18. Down logs in midelevation stand of old growth in the H. J. Andrews Experimental Forest. Logs occupied 29 percent of the forest floor in this stand.

Table 11—A 5-class scheme for rating decomposition of Douglas-fir logs[1]

Characteristic	Decay class 1	2	3	4	5
Bark	intact	mostly intact	partially intact to sloughing	absent	absent
Twigs, 1.2-inch (3-cm)	present	absent	absent	absent	absent
Large branches	present	present	present	present	absent
Exposed wood texture	intact	intact to partly soft	large, hard pieces	small, soft, blocky pieces	soft and powdery (when dry)
Portion of log on ground	support points	support points and slightly sagging	log is sagging	all	all
Exposed wood color	original	original	original to red-brown	light brown to reddish	red-brown to dark brown
Epiphytes	none	none	conifer seedlings (≤3 years old)	moss and hem-lock seedlings	moss and hem-lock seedlings
Log shape	round	round	round	round	oval
Invading roots	none	none	confier seedlings	in sapwood only	in sapwood and heartwood
Characteristics not used that also apply:					
Fungal fruiting bodies	none	*Cyathus, Tremella, Mycena, Collybia, Polyporus, Fomitopsis, Pseudohydnum*	*Polyporus, Polyporellus, Pseudohydnum, Fomitopsis*	*Cortinarius, Mycena, Marasmius*	*Cortinarius, Collybia, Cantharellus*
Mycorrhizae	none	none	none	in sapwood	in sapwood and heartwood

[1] Adapted from Fogel et al. (1972).

Like standing dead trees, logs go through recognizable stages of disintegration. One system for classifying the stage of decay of down logs is a five-class scheme based on easily recognized physical characteristics (table 11); some classes are shown in figure 19. As indicated in figure 10, standing dead trees may directly enter any of the first four log decay classes, depending on their condition when they fall. For example, a live tree, uprooted or broken off in a windstorm, becomes a decay class 1 log, whereas a very rotten snag might collapse into a decay class 3 or 4 log.

Figure 19. These logs are representative of several stages of decay or "decay classes."

Important physical and chemical changes are associated with the progression of decomposition (fig. 20). Logs increase in moisture content at a very early stage in decomposition and retain significant quantities of water thereafter. This is an important factor in their suitability as wildlife habitat and as sites for establishment of tree seedlings and for N fixation. Nitrogen becomes concentrated in logs as decay progresses; over threefold increases occur between class 1 and 5 stages of decay. The concentrations of phosphorus and calcium show patterns similar to N.

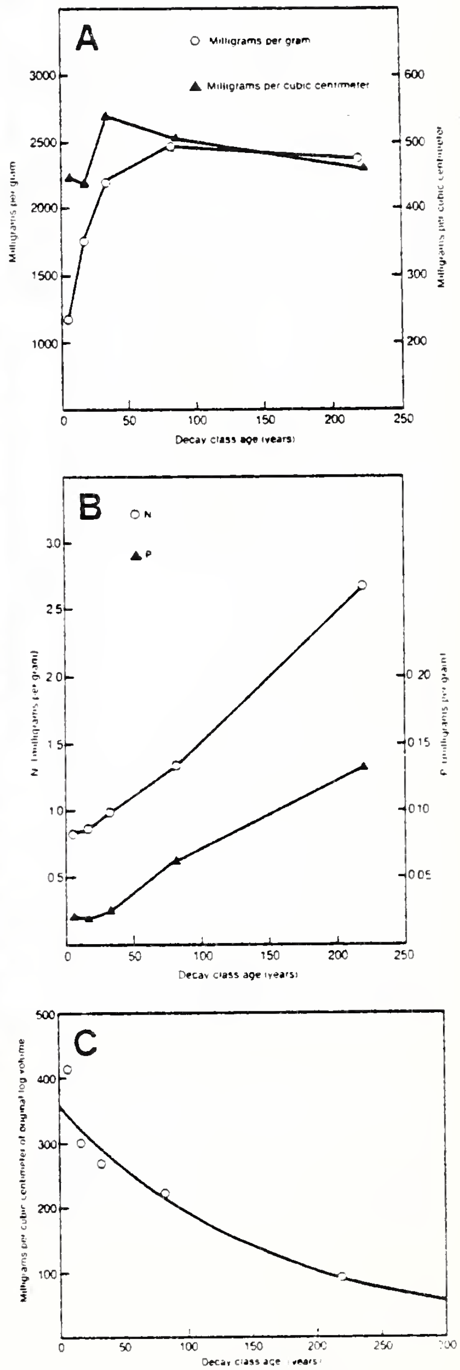

Figure 20. Important physical and chemical changes are associated with the progression of decomposition in logs. A. Changes in percent of water and volume of water per unit of wood with time; note the rapid increase in percent water early in decomposition. B. Changes in content of nitrogen (N) and phosphorus (P) with time. C. Changes in density of wood with time.

Large logs disappear slowly. In one old-growth Douglas-fir stand at mid-elevation, for example, a class 5 log had charcoal on surfaces in contact with mineral soil, suggesting that it had fallen at the time of the wildfire that initiated the present stand—470 years ago! Decomposition rates can be expressed as decreases in log density (fig. 20c). Logs lose only about 40 percent of their original density after 150 to 200 years. Based on decomposition models, 480 to 580 years are estimated for a 30-inch (80-cm) diameter Douglas-fir log to become 90 percent decayed.[12]

Habitat Function. Logs provide essential habitat for a variety of invertebrates (Deyrup 1975) and vertebrates (Maser et al. 1979). They are used as sites for lookouts, feeding and reproduction, protection and cover, sources and storage of food, and bedding. The high moisture content of logs makes them particularly important as habitat for amphibians.

Maser et al. (1979) reported that 178 vertebrates use logs in the Blue Mountains—14 amphibians and reptiles, 115 birds, and 49 mammals; they tabulated use by log decay classes for each species. Logs are considered important in early successional stages as well as in old-growth forests. The persistence of large logs has special importance in providing wildlife with habitat continuity over long periods and through major disturbances.

[12] Means, Joseph E., Kermit Cromack, Jr., Paul E. MacMillan, and Alfred T. Brown. Models of Douglas-fir log decomposition in a 450-year-old forest. 25 p. Unpublished manuscript on file at Forestry Sciences Laboratory, Corvallis, Oreg.

Logs may contribute significantly to reestablishment of animal populations by providing pathways along which small mammals can venture into clearcuts and other bare areas. This has relevance to the reestablishment of tree seedlings on bared areas since survival and growth of new trees depend on development of appropriate mycorrhizal associations. Surprisingly, fungal symbionts apparently disappear from cutover areas shortly after their host trees are removed (Harvey et al. 1976a), and the sites must be reinoculated with their spores. Many mycosymbionts have underground fruiting bodies and completely depend on animals for dissemination of spores. Small mammals are the vectors. They consume the fungus and carry spores to new areas, thereby inoculating tree seedlings (Maser et al. 1978a, 1978b; Trappe and Maser 1978).

Logs also serve as sites for reproduction of tree species, especially western hemlock (fig. 21). This is clearly an important function in natural stands since these seedlings and saplings supply replacements as openings appear in the overstory canopy. In one old-growth stand at midelevation in the Cascade Range, over 64 percent of the western hemlock and 4 percent of the Pacific silver fir reproduction were rooted in rotten wood.[13] The phenomenon of nurse logs is widespread in the forest types of the Pacific Northwest. Minore (1972) found that seedlings of both Sitka spruce and western hemlock were more numerous and taller on rotten logs than on the adjacent forest floor at Cascade Head Experimental Forest in the Sitka Spruce Zone (Franklin and Dyrness 1973) on the Oregon coast. In the South Fork Hoh River Valley of Olympic National Park, also in the Sitka Spruce Zone, reproduction of spruce and hemlock is essentially confined to rotten wood substrates;[14] different species of logs also vary in their suitability as nurse logs as evidenced by differences in densities of seedlings. In subalpine environments, such as the Pacific Silver Fir Zone, successful reproduction of western hemlock is inevitably associated with rotten logs which are also heavily colonized by Pacific silver fir (Thornburgh 1969, Franklin 1966). Rotten logs as seedbeds or "nurseries" may have practical significance in a variety of situations; for example, as sites for natural reproduction after shelterwood or selective cuttings or for planting in cutover areas. Wood substrates appear to have particular silvicultural importance in coastal environments and where reproduction of western hemlock is desired. Rotten logs can also be of key significance in perpetuating campgrounds in old-growth forests in the Cascade and the Coast Ranges by providing seedbeds for tree reproduction.[15]

[13] Means, Joseph E. Personal communication. USDA Forest Service, Forestry Sciences Laboratory, Corvallis, Oreg.

[14] LaRoi, George H., and Jerry F. Franklin. 1979. *Picea sitchensis* and *Tsuga heterophylla* recruitment and survivorship in alluvial rain forests of the Hoh River Valley, Olympic National Park. Unpublished manuscript on file at Forestry Sciences Laboratory, Corvallis, Oreg.

[15] Moehring, Janis, and Jerry F. Franklin. 1977. Ohanapecosh campground rehabilitation study. 51 p. Unpublished report for the National Park Service, Mount Rainier National Park.

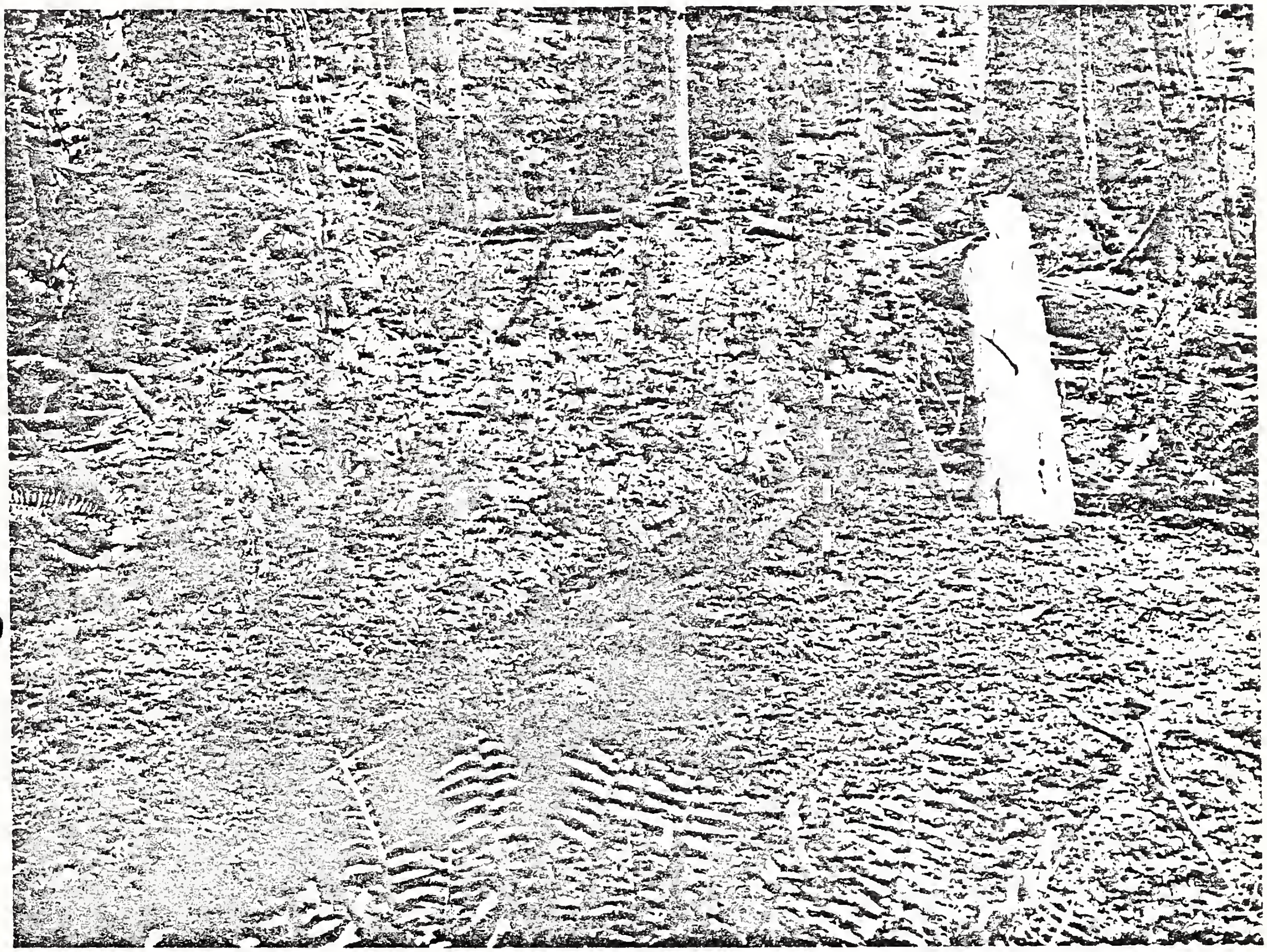

Figure 21. Decaying logs are important habitat for the reproduction of tree species; this "nurse log" has seedlings of western hemlock, western redcedar, and Sitka spruce.

Rotten wood is also critical as substrate for ectomycorrhizal formation. In one coniferous forest stand, over 95 percent of all active mycorrhizae were in organic matter—of which 21 percent were in decayed wood (Harvey et al. 1976b). In another study in the northern Rocky Mountains, decayed wood in soil was important in moist, mesic, and arid habitat types (Harvey et al. 1979); it was the most frequent substrate for active ectomycorrhizae on the dry site, probably because of high moisture levels in the wood. Mycorrhizal fungi can colonize logs, presumably using them as sources of water and nutrients (Harvey et al. 1978). The mycorrhizal relationships may be important factors in establishment of seedlings on nurse logs; they are also important to mature trees.

Just as quality and special properties of wood products vary by tree species, the natural ecological characteristics of logs also vary by species. Average size of logs and slow rates of decay make some species, such as Douglas-fir and redcedar, more persistent. Differences in value of species as nurse logs may relate to physical and chemical properties of the log or the type of wood rot infesting it; the inferior performance of western hemlock in the role of nurse log is one example.

Cycling Function. Based on current knowledge, the most important cycling functions of logs are as sinks or storage compartments for energy and nutrients and as sites for N fixation. In addition, logs may provide physical stability, protecting the site from surface erosion.

The accumulations of carbon and nutrients represented by logs can be very large. In the short-term view, this material is a sink since it is made available so slowly. On the other hand, it is a significant source of stored energy and nutrients and one that can "bridge" major disturbances (can be continuous from an old-growth forest through a wildfire to a young-growth stand). Trunks of live trees, snags, and logs are structures in which N is retained through a wildfire; whereas in more easily burned organic components (leaves, forest floor) substantial N is volatilized and lost from the system.

The discovery of significant bacterial N fixation in coarse woody debris is recent; it occurred almost simultaneously in the forests of the Northeastern, Southeastern, and Northwestern United States. The most thorough study reported to date is Roskoski's (1977) in northeastern hardwood forests. Greater decay and higher moisture contents were associated with a higher incidence of N fixation in woody debris. Fixation occurred in an average of 25 percent of the wood samples >0.4 inch (1 cm) in diameter. Larger woody debris was probably more favorable for N fixation because of better moisture conditions (and consequent low oxygen levels) and because larger pieces last longer, disintegrate more slowly, and, therefore, provide greater opportunity for inoculation by the appropriate bacteria. Larsen et al. (1978) and Roskoski (1977) found higher rates of N fixation in logs at advanced stages of decay and higher moisture content.

Roskoski (1977) estimated total N fixation in hardwood stands of different ages. The largest amounts were in the youngest (14 years old—1.25 ± 1.80 kg/ha per year) and oldest (200 years old—0.96 ± 0.77 kg/ha per year) stands. The amounts of N fixed were directly related to the weight of coarse woody debris which was 17 tons per acre (38 tonnes/ha)—all from the previous old-growth forest—and 15 tons per acre (34 tonnes/ha) in the 4- and 200-year-old stands, respectively.

Amounts of N fixation have not yet been estimated for old-growth coniferous forests in the Pacific Northwest, although N fixation has been detected. Substantially greater amounts are expected than those estimated for eastern hardwood forests, based on the large tonnage of woody debris present (5 to 15 times that reported by Roskoski 1977), and the much larger average size of material in the Douglas-fir region. We estimate approximately 4 pounds per acre (4.4 kg/ha) per year of N fixed in an average old-growth stand;[18] a daily rate from Larsen et al. (1978) of 2.0×10^{-9} moles per gram of logs is assumed in all decay classes throughout the year in the more moderate year-round climate on the west coast.

Decaying logs and decayed wood in soil are also of overwhelming importance as sites for nonsymbiotic N fixation in forests in the northern Rocky Mountains (Harvey et al. 1978); this is especially true of dry sites and during dry periods. Nitrogen fixation may occur in standing dead trees, even with their wider fluctuations in moisture content and aeration, but the amounts are unknown. Fixation is also associated with development of heart rots in live trees (Aho et al. 1974). Harvey et al. (1978) provide examples of both situations—N fixation associated with *Fomes pinicola* (a saprophytic fungus) in dead Douglas-fir and with *Echinodontium tinctorium* (a heart rot) in live western hemlock.

Logs have numerous important roles in cycling nutrients and maintaining site productivity. Complete removal of wood residues by harvesting and broadcast burning eliminates logs as reservoirs of nutrients and water and as entities whose physical effects may have a positive influence on the quality of a site (Jurgenson et al. 1977). Increased commercial use of wood residues or complete yarding of unmerchantable residues reduces the amount of sound or partially decayed logs. Since the products of decay and fire (humus, decayed wood, and charcoal) are important to quality and function of forest soils (Harvey et al. 1978, Jenny 1980), future forest management must consider logs as "parent material" for soil organic matter.

[18] Our typical old-growth stand has 65 tons per acre (145 tonnes/ha) of logs and 20 tons per acre (45 tonnes/ha) of snags.

Logs in Streams. Large, woody debris in streams is a dominant element in aquatic ecosystems of old-growth forests. Debris largely controls the distribution of aquatic habitats, stability of streambeds and streambanks, and routing of sediments and water through the stream system.

Large, organic debris enters streams by a variety of mechanisms, some of which are interrelated and occur as a chain reaction. Principal mechanisms of input are blowdown, slides, avalanches, deep-seated mass movements from adjacent slopes, and undercutting of streambanks. Large debris is exported from sections of streams by flotation at high streamflow; in massive debris torrents involving the rapid down-channel movement of a slurry of soil, alluvium, and large organic debris; and by movement of fine particulates and dissolved material produced by biological and physical breakdown of logs.

The general character of large debris in small (first to second order) and intermediate-size (third to fourth order) streams in old-growth Douglas-fir is shown in figures 22 and 23. The first-order stream (fig. 22) is choked with debris, located essentially where it fell. The third-order stream (fig. 23) is large enough to float and redistribute some of the debris, forming accumulations along the channel. The third-order stream is still narrow relative to the length of individual pieces of debris, so accumulations of debris affect the width of the entire channel. In fifth-order and larger rivers, debris from old-growth forests accumulates high on streambanks at high flow, and, therefore, plays only a minor direct role in the physical and biological character of the river (Keller and Swanson 1979).

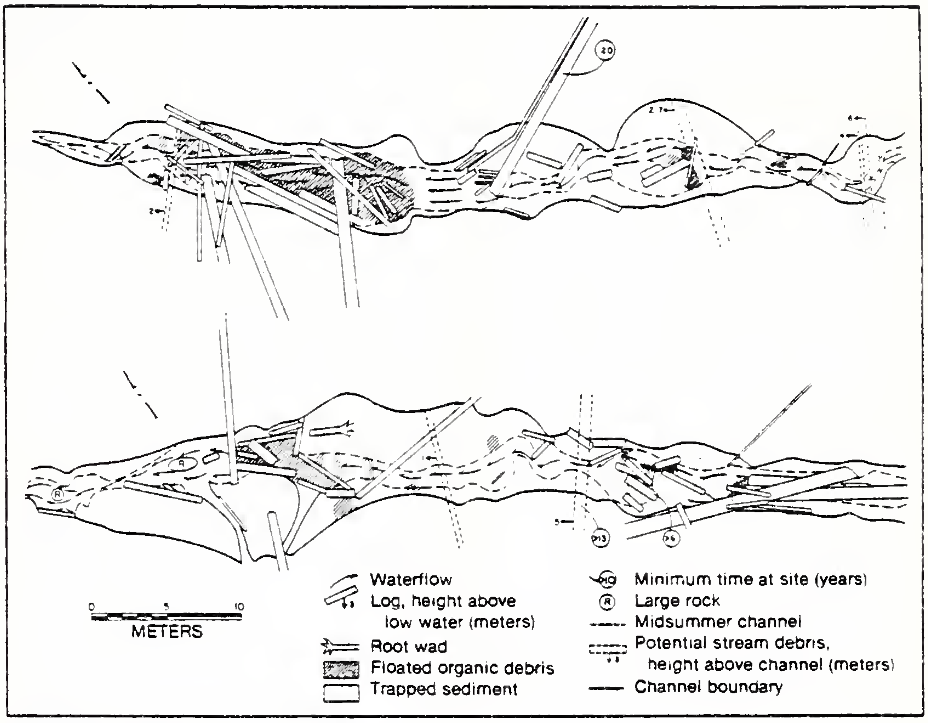

Figure 22. Small, first-order streams in old-growth forests are often choked with coarse woody debris as in these mapped sections of a stream in the H. J. Andrews Experimental Forest, western Oregon Cascade Range.

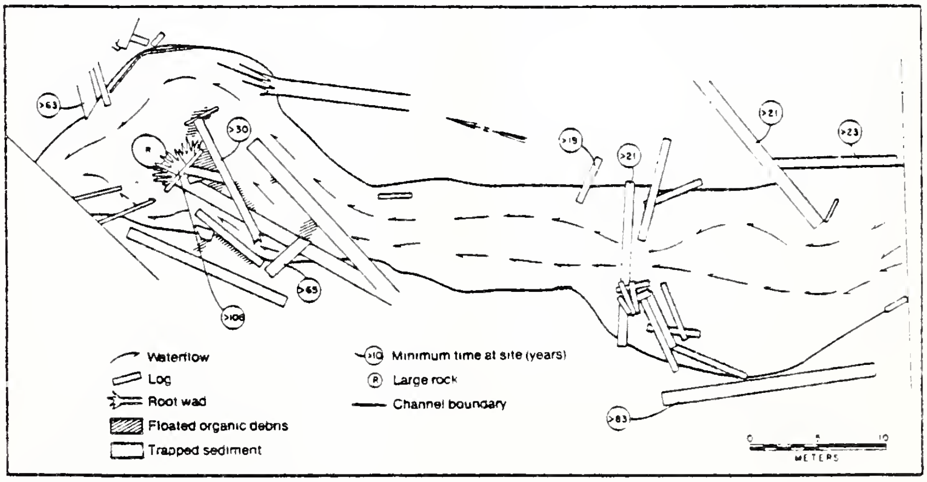

Figure 23. Third-order streams are large enough to float and redistribute much of the woody debris, forming distinct accumulations as in this section of Mack Creek in the H. J. Andrews Experimental Forest, western Oregon Cascade Range.

Table 12—Loading of coarse (>4-inch or 10-cm diameter) debris in sections of 5 streams flowing through old-growth Douglas-fir forests in the McKenzie River system, western Oregon[1]

Stream	Loading of coarse debris		Length of sample area		Width of channel		Gradient of channel	Stream order	Watershed area	
	Pounds per square foot	Kilograms per square meter	Feet	Meters	Feet	Meters	Percent		Square miles	Square kilo-meters
Devilsclub Creek	8.7	43.5	297	90	3.3	1.0	40	1	0.08	0.2
Watershed 2 Creek	7.6	38.0	445	135	8.6	2.6	26	2	.32	.8
Mack Creek	5.7	28.5	990	300	40	12	13	3	2.4	6.0
Lookout Creek	2.3	11.6	990	300	79	24	3	5	24.2	60.5
McKenzie River (Rainbow)	.1	.5	2,640	800	130	40	.6	6	410	1 020

[1] Specific gravity of wood is assumed to be 0.50 gram per cubic centimeter.

The relationship between size of debris and size of stream controls the arrangement and concentration of large debris at different points in a river system. In a downstream sequence, conditions change from (1) abundant, randomly distributed debris to (2) moderate loading of debris in a clumped distribution, and finally, to (3) a low concentration of widely distributed debris. The general decrease in loading of coarse debris downstream is exemplified in the McKenzie River system where headwater tributaries may have nearly 10 pounds per square foot (50 kg/m²) of coarse woody debris, whereas the mainstream McKenzie River at Rainbow has only 0.1 pound per square foot (0.5 kg/m²) (table 12).

Quantities of large debris in streams flowing through old-growth forest are large and variable. Froehlich (1973) measured large organic debris (>10-cm or 4-inch diameter and >0.3-m or 1-foot length) in 10 relatively undisturbed streams draining from 6 to 120 acres (2.4 to 49 ha). Values ranged from 7.1 to 24.8 tons per 100 feet (21 to 73.9 tonnes/100 m); mean was 13.8 tons per 100 feet (41.1 tonnes/100 m) and standard deviation, 6.6 (19.7). In five streams draining 282 to 1,593 acres (114 to 645 ha), loadings of large debris were 5.7 to 14.2 tons per 100 feet (17 to 43 tonnes/100 m); mean was 10.3 tons per 100 feet (30.7 tonnes/m) and standard deviation, 3.6 (10.7).

Expressed on an area basis, concentration of large debris averaged 10.1 pounds per square foot (50.6 kg/m²) in the small streams draining less than 120 acres (48 ha) of old growth. In contrast, large debris in four natural, second-growth stands initiated by wildfire 75 to 135 years ago averaged only 3.8 pounds per square foot (19.1 kg/m²). Furthermore, 75 percent of the coarse debris loading in these streams was made up of large diameter, decomposed pieces derived from the prefire, old-growth stand. Structural elements inherited from preexisting old-growth stands may persist in a stream through much of the development of second growth. Swanson and Lienkaemper (1978) hypothesized that streams flowing through managed, short-rotation forests will have very low loadings of coarse debris because management activities, such as harvesting and thinning, remove the source of large debris.

Loading of debris may be abruptly reduced by debris torrents, or "sluice-outs." Debris torrents are rare occurrences in old-growth ecosystems; the return period of torrents in a 25-acre (10-ha), old-growth watershed in the H. J. Andrews Experimental Forest has been estimated to be from 500 to 600 years. Torrents may reduce loading of debris to less than 1.0 ton per 100 feet (3.0 tonnes/100 m) of stream. As torrents move past forested streamside areas, however, banks are undercut, destabilizing trees and leading to a several-decade post-torrent period of relatively high inputs of coarse debris and reestablishment of stable debris dams and associated stream habitat. Where streamside stands are very young (less than 30 or 40 years), pieces of debris after a torrent may not be large enough to form stable accumulations and associated habitat.

The history of large debris in streams has been examined by dendrochronological methods (Swanson et al. 1976, Swanson and Lienkaemper 1978). Large pieces of debris commonly have been in streams for 25 to more than 100 years and have, therefore, weathered extreme events, such as the December 1964 flood in the Pacific Northwest. Western redcedar is particularly long lasting, followed, in order of increasing rate of breakdown, by Douglas-fir, western hemlock, and riparian hardwoods. This long residence time results from both the characteristic predominance of slowly decomposing Douglas-fir and western redcedar and the large debris provided by old-growth stands.

Large debris in streams strongly influences morphology of the channel and routing of sediment and water. In first-order through about third-order streams, debris helps to form a stepped longitudinal stream profile. Much of the energy of a stream is dissipated at falls or cascades created by debris. This pattern of dissipation in a small proportion of total stream length results in less energy available for erosion of bed and banks, more storage of sediment in the channel, slower routing of organic detritus, and greater diversity of habitat than in straight, even-gradient channels.

In small and intermediate streams in the Pacific Northwest, large debris may be the principal factor determining the characteristics of aquatic habitats. The important role of debris in creating habitat for fish has been reviewed by Narver (1971), Hall and Baker (1975), and others. The wood itself is a habitat or substrate for much biological activity by microbial, invertebrate, and vertebrate organisms.

The general influence of woody debris on aquatic habitat has been measured in several streams in the H. J. Andrews Experimental Forest (Swanson and Lienkaemper 1978). In an 800-foot (240-m) section of Mack Creek, which flows through old growth, 11 percent of the stream area is woody debris, 16 percent is wood-created habitat (primarily depositions of sediments behind woody debris), and 73 percent is nonwoody habitat—mainly cascades dominated by boulders. This section occurs in a third-order stream channel draining about 1,500 acres (600 ha). Woody debris occupies 25 percent and wood-created habitat another 21 percent of the stream area of Devilsclub Creek, a first-order tributary of Mack Creek draining 25 acres (10 ha). Much of the biological activity in the processing of detritus and other consumer organisms is concentrated in the wood and wood-created habitats.

Over half the inputs of N are associated with woody debris, including N fixation on wood. Quantitatively, woody debris is also the major energy input.

Distinctions between stream ecosystems in old-growth and second-growth stands are less clear than those between stream orders. Large organic debris plays the dominant role in streams in old-growth forests and clearly does not in managed second-growth forests. Natural, second-growth stands have residual debris dams as well as residual large stems for new dams that help span the period until the young stand begins producing larger diameter woody debris. Analysis of the overall response of aquatic communities to long-term changes in structure and nutrient sources, such as those resulting from reduced concentrations of large, organic debris, is an important research problem.

In summary, coarse woody debris is extremely important to streams in old-growth coniferous forests. Debris dams and associated plunge pools and trapped sediments, such as gravel bars, provide a great diversity of habitats for organisms. The resulting stepped stream profiles provide for greater physical and biological stability by dissipating energy otherwise used in cutting channels and moving sediment. Debris dams also slow the routing of other organic inputs, allowing organisms time to more fully process these materials before they are exported downstream. Finally, the woody debris is itself a major source of energy and nutrients for the stream ecosystem.

Old-growth forests provide highly specialized habitats and are neither decadent, unproductive ecosystems nor biological deserts. In this paper, we have contrasted the compositional, functional, and structural features of old growth with those of young-growth forests wherever data permit. It is clear that there are major contrasts between old-growth and managed, young-growth stands; structures, species, and patterns and rates of material (carbon and nutrient) flows and cycles characteristic of old growth will be absent from managed lands unless specifically provided for in silvicultural prescriptions. Differences between old-growth stands and natural, second-growth stands often appear to be one of degree; however, this is often because of material carried over from the old to the new stand, such as large snags and logs, which provide woody debris and animal habitat and serve other functions.

Alternatives for managing old-growth forest ecosystems or attributes include management to: (1) perpetuate existing old-growth stands, (2) re-create ecosystems with old-growth characteristics by long rotations, and (3) provide for individual features of old-growth forests. The first two alternatives involve a relatively small percentage of the commercial forest land in the Pacific Northwest; various acreages have been proposed in Federal land-use planning exercises, including spotted owl management plans, generally totaling 5 percent or less of the land base. Perpetuation of existing stands is the surest course since the old-growth conditions exist and can be expected to persist for several additional centuries. This assumes viable sites (appropriate size and shape with boundaries that can be protected) are selected and catastrophic fire and windthrow are avoided. Re-creation of old-growth conditions through long rotations is theoretically possible but clearly unproved at this time. Site selection is important, though perhaps not as critical as it is where perpetuation of an existing old-growth forest is the objective.

The third alternative—providing for individual features—involves applying the information from this report to the main body of commercial forest land in the Pacific Northwest, not simply to the small enclaves managed for old growth. Our knowledge of old-growth forests and how they work can be used to advantage in managing millions of acres of forest for timber production by improving their ability to provide additional ecological benefits. Some of the most important ecological features of old-growth forests, and, indeed, the entire natural forest sere, can be duplicated on these lands with relatively small sacrifice of production of timber. Knowledge of old growth, such as that about snags and coarse woody debris, is widely applicable to forest management.

If the forest manager decides to retain, maintain, or re-create some forest ecosystems with old-growth characteristics several questions require attention. How should the old-growth allocation be distributed over the landscape? Which management practices should be followed in areas selected for retention of old growth and on areas to be managed on long rotations? Size and shape of old-growth enclaves are distributional concerns. Structural features are the management key in either perpetuating or re-creating old-growth stands.

Distribution of Old-Growth Management Areas. Having decided to maintain a certain percentage of a management unit as old-growth forest, a land manager must decide how it is most effectively distributed. Should a small stand be maintained in each drainage or compartment or fewer large tracts reserved? Where are boundaries best located? The history of natural wildfires is somewhat instructive, at least in indicating the landscape patterns in which existing old-growth stands originated. Stands were typically established in blocks of hundreds of acres. Boundaries most often were along topographic features, such as ridges or streams. "Feathering" of boundaries, resulting in large areas of mixed stands (residual old growth scattered through a young stand), was common. Fires often skipped large patches of trees, particularly on lower slopes and stream bottoms, or stands protected by natural barriers.

How large a drainage unit or stand is essential for a viable old-growth ecosystem is a difficult question. Generally, an area of 300 to 500 acres (120 to 200 ha) is sufficient for most plants and animals. This size has been suggested as essential to a pair of breeding northern spotted owls, for example, one of the most wide-ranging species dependent on old growth. McClelland et al. (1979)

suggest 50 to 100 acres (20 to 40 ha) for the nesting and feeding areas of cavity-dwelling birds. Generally, 500 to 1,000 acres (200 to 400 ha) are needed for a third-order stream drainage. Areas much smaller than 500 acres (200 ha) can, on the other hand, preserve many attributes of old-growth forests, particularly if the boundaries chosen for the area prevent rapid deterioration of the stand.

Entire drainage basins, even for small first- or second-order streams, are ecologically the most desirable units for old-growth forest management. Such drainages have natural topographic boundaries that often provide superior protection from windthrow and other external influences for the reserved stand. A variety of ecological conditions are present, along with a stream system in which natural land-water interactions (for example, woody debris inputs to the stream) can continue. Plant and animal diversity will also be higher in a drainage basin than in an isolated upland forest stand of the same size.

Another useful location for old-growth management areas is along streams. Streamside strips (buffer zones) were originally conceived for shading streams and minimizing increases in water temperatures; a more valuable function in streams up through at least third order is in providing essential energy, structural inputs (for debris dams), and stability of banks. Debris dams must be continually created to replace broken dams; this is particularly important after infrequent debris torrents that remove all, or most, pieces of large debris. Streamside strips of old-growth forest will provide for continued physical and biological stability of the aquatic ecosystem.

Streamside and roadside strips of old growth have the additional advantage of providing migration routes for organisms dependent on mature forests. The strips skirt managed stands and provide continuity between otherwise isolated pockets of natural and old-growth forest. Such migration routes may be important for avoiding loss of species as "islands" of habitat suited to a species become more limited and isolated (MacClintock et al. 1977). The protected travel routes provided by these strips allow organisms to migrate in response to shifts in location of suitable habitat.

Reserve strips must be in appropriate locations and of sufficient size to survive normal windstorms. Streamside strips have often been narrow (since shading the stream was the primary objective) and sharp edged; consequently, they are extremely susceptible to blowdown. The wetter soils on lower slopes and streamsides do not make retention of strips any easier, and upslope harvesting often intensifies this problem. Regular programs to salvage logs on roadside or streamside strips are inappropriate because they accelerate deterioration of the stand and remove the essential structural components. Retention of dead wood is especially important for streams where it is the source of stabilizing debris dams. Salvage logging may be appropriate if losses from catastrophic windthrow or other causes occur. Even then, the salvage should not be complete; some down material should be selected and left on the land and in the stream. A 200-foot-wide (60-m) streamside or roadside strip is not a viable unit in most cases; considerable ingenuity and effort will be necessary to identify and lay out viable reserve strips.

Areas with problems that limit potential for management may also be appropriate sites for perpetuating old-growth ecosystems and organisms. An example could be steep landslide-prone headwall areas that depend on a strong and continuing root mantle for stability.

Structural Attributes in Perpetuating or Re-creating Old Growth. The distinctive structural features—the large, old-growth trees, snags, and logs on land and in streams—provide the major key to management strategies. The unique, important compositional and functional features of an old-growth forest usually accompany these structural elements.

An old-growth forest is much more than simply a collection of large trees. The dead, organic component is as important as the highly individualistic, large trees. Decaying snags and logs, particularly in streams, are beneficial and must be provided for in management schemes; they should not be viewed solely as waste, fire hazards, or impediments to management. Snags and logs play important roles as habitat for various organisms and in conserving and cycling nutrients and energy. To a large degree, success in managing forests for old-growth attributes will depend on learning to manage the dead, organic material as cleverly as the live trees.

There are implications here for management of old-growth stands selected for perpetuation. Salvage logging is inappropriate since it removes at least two of the major structural components—dead and down—that are key elements of the system. In all likelihood, some of the more decadent, live trees would also be removed. Salvage logging is also inappropriate because of the damage inevitably done to root systems and trunks of the residual stand which results in accelerated mortality of trees and overall deterioration of the stand.[17]

[17] When stands are selected for preservation (for example, as roadside or streamside strips), the first (and frequently repeated) management activity is often a salvage program. If a manager wants to retain an old-growth ecosystem or a mature forest stand, entries should be avoided or at least minimized. Trees viewed as safety or fire hazards may better be felled and left in place than removed.

There are also implications if the manager wishes to create an old-growth environment (after a stand is cut) by using a long rotation. Initially, foresters may retain larger amounts of woody residues, especially down logs, from the previous stand. Retention of scattered individual, old-growth trees may be useful as sources of epiphytic flora and eventually of large, dead standing trees and down logs.

Rapid development of large, long-crowned trees as early as possible is a key objective of management that can be aided several ways. Selection of understocked stands of reproduction as sites for creating old-growth stands is one approach since individuals will grow faster and lose lower branches more slowly under open-grown conditions. Many existing old-growth stands may have regenerated slowly (Franklin and Waring 1980); growth patterns of individual trees suggest growing conditions essentially free from competition for a century or more. If initial densities of stands are moderate—at current recommended levels for managed stands—precommercial and commercial thinnings will be necessary during the first 100 years of a long-rotation forest management cycle. Growth rates of individual trees will be too low at high densities, or at moderate densities on less productive sites, to produce desired sizes of stems even after 200 years; thinnings and partial cuttings are essential under those conditions. Great care must be taken, however, to minimize damage to residual trees.

Creation of appropriate types and amounts of standing dead and down trees is a specific management objective. Snags and logs from the original stand should be avoided during intermediate cuttings. Up to about 100 years, the size of snags and logs produced by the young stand is probably not of particular ecological importance. Some of this material could be removed along with excess live trees—those that will die before reaching significant diameter (50 cm or 20 inches). Openings for development of shade-tolerant species can also be created this way; if these species do not come in naturally, they could be artificially introduced, possibly by underplanting. The large snags, logs, and any live old growth left from the original stand should not be removed during salvage operations.

After about 100 years, partial cutting of any type becomes increasingly inappropriate. There are fewer live dominants, and their loss, either directly by cutting or gradually through damage to roots and trunks, is undesirable. Standing dead trees and logs now being recruited from the live stands are of sufficient size to fully perform desired habitat and cycling functions.

To summarize, if the objective is perpetuation of an old-growth forest ecosystem, a minimum amount of disturbance should be allowed. Snags and logs perform important functions and are essential structures. When the objective is to create an old-growth forest from scratch, large individual trees with large crowns should be grown as quickly as possible. Scattered old-growth trees and rotten logs from the original stand should be retained and reproduction of western hemlock, western redcedar, and other shade-tolerant associates under the Douglas-fir canopies encouraged. Partial cuttings may be useful and will be necessary in moderate to heavily stocked stands of reproduction if large trees are to be attained as quickly as possible. After trees are about 100 years old, such cuttings are increasingly inappropriate, however.

For multiple-use objectives, an increased awareness of the nature and nontimber value of individual trees is important; for example, potential or current value as habitat for epiphytic communities and wildlife. Knowledge of the ecological roles of standing, dead trees and logs beyond their value as wildlife habitat is also desirable.

Managing for Individual Old-Growth Attributes

There is considerable logic in managing entire stands or small drainages for old-growth attributes. The old-growth ecosystem is a system of many interlinked components, including organisms. The serial relationship of the key structural components has been discussed—from a large, old-growth tree to a nearly decomposed, rotten log (fig. 10). Further, some organisms or functions may depend on an intact old-growth forest for their perpetuation.

Nevertheless, a forester may wish to manage for some individual old-growth attributes. This is, in fact, how the forester can put some of the information on old growth to work to increase ecological benefits from intensively managed timberlands. The structural components again provide the key. Perhaps most obvious is providing for large snags and logs. This can be done in the first-generation managed forest by retaining some of this material from the virgin stand. The tendency has been to remove all such materials as a safety measure and to reduce logging residues, which are viewed as fire hazards and impediments in regeneration and other silvicultural activities. In second- and third-generation stands, a forester will have to create appropriate materials since neither large snags nor logs will usually be present.

The need for snags was recognized first by wildlife managers, and they have more recently recognized the value of logs (Maser et al. 1979, Thomas et al. 1979a). Thomas et al. (1979a) led in developing guidelines on sizes and numbers of such material needed to provide for vertebrates; although their research was conducted in the Blue Mountains of eastern Oregon and Washington, the same principles apply on the west side of the Cascade Range, as shown by Mannan et al. (1980) and Cline et al. (1980). These authors suggest that snags be created from defective, living trees and urge maintenance of large snags covering the spectrum of decomposition. Densities of snags in natural, old-growth stands are proposed as an interim management guide until more data are developed. Cline et al. (1980) also suggest leaving snags in groups to reduce problems of safety and fire control. It is important to remember that much more than habitat for vertebrate animals is involved in preserving snags; standing dead trees and logs serve other functions as well.[18]

There are currently no good guides to the number and sizes of logs that should be left on cutover areas. Removal of all coarse woody debris is not the best ecological practice. Costs and benefits of some practices (such as yarding unmerchantable material) are not known; negative impacts on long-term site productivity, wildlife, and erosion may offset the benefits to fire protection and ground accessibility. It does appear that at least several larger logs per acre are needed for wildlife, especially small mammals. Defining the types and sizes of logs and other woody debris desired in managed stands is a major problem for research.

Retention of small groups of old-growth trees, or scattered individual trees, may be a useful practice. This was conceived as a technique for providing a source of epiphytic "inoculum" for adjacent young trees. Lack of such a source may be a factor in the absence of the N-fixing epiphytes on trees less than 150 years old. Leaving occasional old-growth trees has another advantage—it will, in the long run, provide a source of large snags and logs. This may be the easiest strategy for perpetuating these structural components into second- and even third-generation managed stands.

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[18] The role of dead wood in cycling and conserving nutrients, especially N, is an outstanding example. Ten years ago nothing was known about sources of N in old-growth stands, other than the atmospheric input. In the interim, epiphytic lichens and wood-dwelling bacteria have been identified as significant sites for fixation. There are several important sources for additions of N in both the early stages of forest succession—nonleguminous N fixers, such as alder (*Alnus* spp.) and ceanothus (*Ceanothus* spp.)—and in old-growth forests. Existing management strategies call for quick establishment of conifer canopies and short rotations—which effectively eliminate these additions of N to intensively managed sites.

Trees Mentioned in This Report

Common name	Scientific name
Alaska-cedar	*Chamaecyparis nootkatensis* (D. Don) Spach
Coast redwood	*Sequoia sempervirens* (D. Don) Endl.
Douglas-fir	*Pseudotsuga menziesii* (Mirb.) Franco
Grand fir	*Abies grandis* (Dougl. ex D. Don) Lindl.
Incense-cedar	*Libocedrus decurrens* Torr.
Noble fir	*Abies procera* Rehd.
Pacific silver fir	*Abies amabilis* (Dougl.) Forbes
Port-Orford-cedar	*Chamaecyparis lawsoniana* (A. Murr.) Parl.
Sitka spruce	*Picea sitchensis* (Bong.) Carr
Sugar pine	*Pinus lambertiana* Dougl.
Western hemlock	*Tsuga heterophylla* (Raf.) Sarg.
Western redcedar	*Thuja plicata* Donn
Western white pine	*Pinus monticola* Dougl. ex D. Don

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Old-growth coniferous forests differ significantly from young-growth forests in species composition, function (rate and paths of energy flow and nutrient and water cycling), and structure. Most differences can be related to four key structural components of old growth: large live trees, large snags, large logs on land, and large logs in streams. Foresters wishing to maintain old-growth forest ecosystems can key management schemes to these structural components.

Keywords: Ecosystems, old-growth stands, stand composition, stand structure, Douglas-fir, *Pseudotsuga menziesii*, western hemlock, *Tsuga heterophylla*.

The **Forest Service** of the U.S. Department of Agriculture is dedicated to the principle of multiple use management of the Nation's forest resources for sustained yields of wood, water, forage, wildlife, and recreation. Through forestry research, cooperation with the States and private forest owners, and management of the National Forests and National Grasslands, it strives — as directed by Congress — to provide increasingly greater service to a growing Nation.

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8

Material Transfer in a Western Oregon Forested Watershed

F. J. Swanson, R. L. Fredriksen, and *F. M. McCorison*

INTRODUCTION

Abiotic transfer of organic and inorganic materials by a diverse family of processes is an essential part of all natural, large-scale ecosystems. Physical processes of material transfer are particularly important in the coniferous forest biome, which contains many geologically youthful and geomorphically active landscapes. High-relief, steep hillslope and channel gradients, dense vegetation, massive trees, and heavy precipitation result in a complex relationship among material transfer processes and vegetation.

In a strict sense, material transfer involves erosion, transport, and deposition. This is equivalent to current usage of the term "sedimentation," which geologists and engineers use to describe transfer of predominantly inorganic material. In a system with significant depositional sites, material transfer includes routing of material through a variety of storage compartments. In this study, which deals mainly with a small, steep watershed where storage opportunities are limited, we emphasize annual material transfer rates and roles of vegetation but do not attempt to quantify deposition and storage.

Material transfer has several important roles in the functioning of forest-stream ecosystems. It is an important mechanism for nutrient redistribution and particularly nutrient export from ecosystems. Erosion and deposition create landforms that offer contrasting habitat opportunities for terrestrial and aquatic organisms on a variety of temporal and spatial scales. Erosion may also determine rates and patterns of succession following or during either erosion disturbances (for example, landslide) or disturbance of vegetation alone (for example, wildfire or insect infestation). These effects may be localized to the scales of root-throw mounds and landslide scars (generally $<2000\ m^2$) or they may extend over broad areas covering many hectares.

There is also a variety of ways in which vegetation regulates rates of erosion processes. These influences of vegetation may result in reduced erosion by the effects of ground cover and rooting strength, or in increased erosion, as in the case of trees serving as a medium for transfer of wind stress to the soil mantle.

The great temporal and spatial variability of erosion processes operating on a single landscape and their complex relationships with vegetation have discouraged attempts to quantify erosion on a process-by-process basis in temperate forest ecosystems. Most comprehensive erosion research has been restricted to semiarid lands (Leopold et al. 1966) and alpine and subalpine environments (Jäckli 1957; Rapp 1960; Benedict 1970; Marchand 1971, 1974, Caine 1976). Temperate forest geomorphology was studied on a broad scale in the central Appalachians (Hack and Goodlett 1960), in the Redwood Creek basin, northern California (Janda et al. 1975), and in a drainage basin on the Oregon coast (Dietrich and Dunne 1978). Numerous studies have dealt with material transfer at the scales of individual processes and small watersheds. Recent work on material transfer in forest ecosystems has centered on elemental and particulate matter input/output budgets for small watersheds (Bormann et al. 1969, 1974; Cleaves et al. 1970; Fredriksen 1970, 1971, 1972, 1975; Likens et al. 1977). In none of these small watershed studies was material transfer examined at the process level over an entire watershed.

The purpose of this chapter is to describe the nature of material transfer in a coniferous forest stream ecosystem in terms of: (1) characteristics of the important transfer processes; (2) relations among them; (3) transfer process/vegetation relations; (4) the relative importance of individual processes and process groupings; and (5) the effects of vegetation disturbance on material transfer in a historical context.

DEFINITION OF PROCESSES

Material transfer processes operating in a watershed are broadly grouped into those affecting hillslopes and those operating in stream channels (Fig. 8.1). The hillslope processes supply dissolved and particulate organic and inorganic material to the channel, where channel processes take over to break down the material and transport it downstream and out of the watershed. Significant transfer processes both on hillslopes and in channels include infrequent, localized, high-magnitude events and continuous, widely distributed, low-magnitude processes (Table 8.1).

Hillslope Processes

Solution transport results from leaching from vegetation, soil, and weathering bedrock or input from atmospheric sources and occurs in the dissolved state in subsurface water or, rarely, as overland flow. *Litterfall* is the transfer of organic matter as the final step in the sequence of events: nutrient uptake by roots, translocation to and incorporation in aboveground biomass, abscission

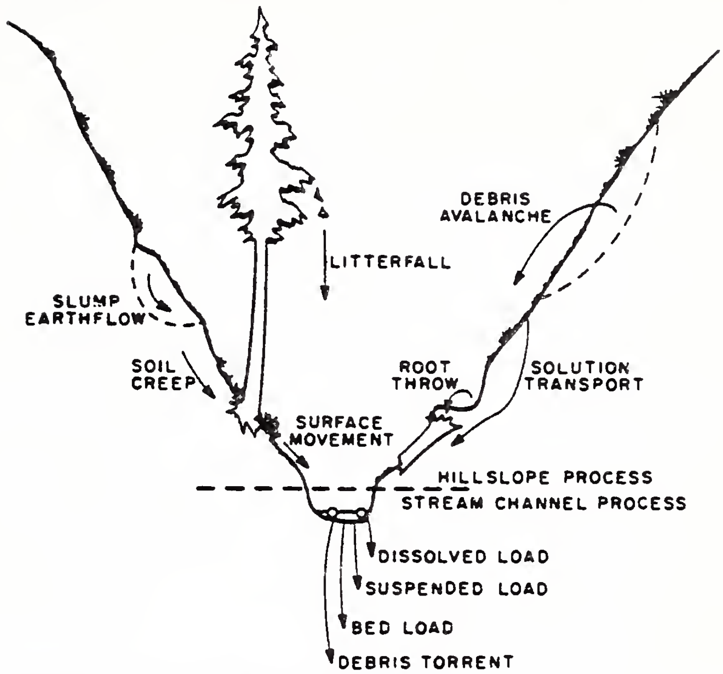

FIGURE 8.1 *Processes that transfer organic and inorganic material in a steep forest watershed ecosystem.*

or breakage, litterfall to the forest floor or stream. In steep terrain, downhill lean of large trees and canopy closure over small streams result in a net downslope displacement of organic matter.

Surface erosion is the particle-by-particle transfer of material over the ground surface by overland flow, raindrop impact, and ice- and snow-induced particle movement and dry ravel, which occurs during dry periods (Anderson et al. 1959). *Creep* is here considered "continuous" creep (Terzaghi 1950); that is, slow, downslope deformation of soil and weathered bedrock. This is a more restrictive definition than that applied by Leopold et al. (1964) and others who include root throw, needle ice, and other processes as part of creep. *Root throw* occurs as movement of organic and inorganic matter by the uprooting and downhill sliding of trees. *Debris avalanches* are rapid, shallow (generally one- to two-meter soil depth) soil mass movements. *Slump and earthflow* are slow, deep-seated (generally five- to ten-meter depth to failure plane) rotational (slump) and translational (earthflow) displacements of soil, rock, and covering vegetation.

TABLE 8.1 *Material transfer process characteristics for watershed 10 in old-growth forest condition.*

Process	Downslope movement rate[a]	Frequency	Watershed area influenced	Landforms
Hillslope processes				
Solution	1	continuous	total	
Litterfall	1	continuous, seasonal	total	
Surface erosion	1	continuous	total	small terracettes
Creep	2	seasonal	total	
Root throw	3	~ 1/yr	0.10%[b]	pit & mound topography
Debris avalanche	4	~ 1/370 yr	1 to 2%[b]	shallow, linear downslope depressions
Slump/earthflow	5[c]	seasonal[c]	5 to 8%	scraps, benches
Channel processes				
Solution	3	continuous	1%	
Suspension	3	continuous, storm	~ 1%	
Bedload	3	storm	~ 1%	channel bedforms
Debris torrent	4	~ 1/580 yr	~ 1%	incised, U-shaped channel cross section

[a]1 = cm to m/yr, 2 = mm/yr, 3 = m/s, 4 = 10 m/s, 5 = mm to cm/yr.
[b]Area influenced by one event.
[c]Inactive in past century in watershed 10.

Channel Processes

Solution transport is movement of material dissolved in stream water. *Suspended sediment transport* is movement of material in colloidal to sand size carried in suspension in flowing water. *Bedload transport* is movement of material approximately coarse sand size and larger by tractive forces imparted by streamflow. *Debris torrent* is the rapid, turbulent movement down stream channels of masses that may exceed 10,000 m^3 of soil, alluvium, and living and dead organic matter. Whole trees may be included. *Streambank erosion* occurs as lateral cutting by a stream as it entrains material such as older alluvium or colluvium moved to the streamside area by creep, surface erosion, or other processes.

Relations Among Processes

The movement of a single particle of material through a watershed is accomplished by a series of steps involving numerous material transfer process-

es. There are in-series, or chain-reaction, relations among processes and there may be superposition of processes operating simultaneously on a particular piece of material. Principal driving variables and sequential relations among erosion processes are shown in Figure 8.2.

Possible types of sequential interactions are varied. Surface erosion rate may be increased by other processes such as root throw and debris avalanche, which expose bare mineral soil and/or locally increase slope steepness. Debris avalanches may be triggered by root throw, and debris avalanche probability is increased by local slope steepening in response to creep, slump, and earthflow activity (Figure 8.2). Probability of root throw is increased by the tipping of trees by creep, slump, and earthflow activity. Creep may be a precursor of debris avalanche, slump, and earthflow movement (Terzaghi 1950), because when strain by creep deformation exceeds a threshold value, discrete, macroscopic failure occurs and translocational or rotational displacement begins.

Hillslope processes supply material to the channel, making it available for transport by channel processes. In the case of debris torrents the debris avalanche, a hillslope process, is a principal triggering mechanism of the channel process. Within the stream environment, chemical and physical processes break down larger particles to smaller ones, and thereby change the relative importance of bed, suspended, and dissolved modes of transport.

In addition to sequential relations, processes work together. A typical column of soil on a steep, forested slope is likely to experience surface erosion, creep, solution transport, and nutrient uptake/litterfall processes simultaneously. The same block of soil may also be subject to slump or earthflow movement.

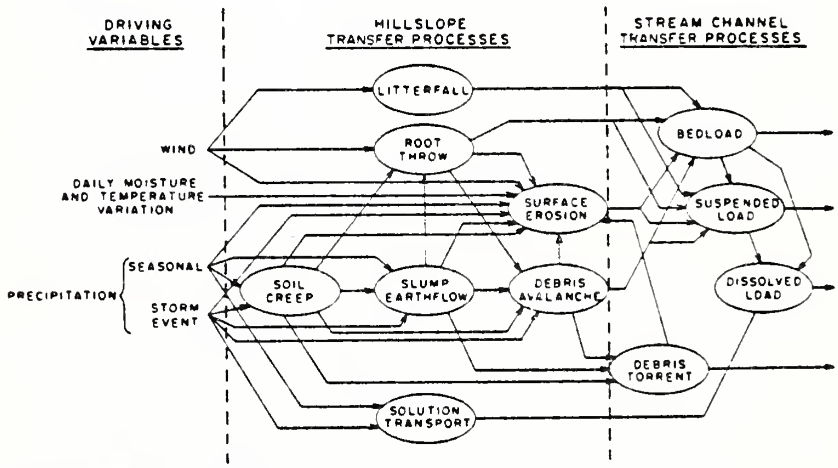

FIGURE 8.2 *Relations among mass transfer processes and principal driving variables. Arrows indicate that one process influences another by supplying material for transfer or creating instability that culminates in the occurrence of the second process.*

Influences of Vegetation on Material Transfer

Vegetation factors regulate rates of transfer processes in a variety of ways (Table 8.2). These factors may either increase or decrease the rate of continuous processes or the probability of episodic events. Knowledge of vegetation-transfer process relations ranges from the obvious (for example, aboveground biomass is the source of litterfall) to the speculative (for example, the effect of windshaking of trees on creep rate). Relations in Table 8.2 are a summary based on inference, direct field observation, modeling studies, and experimentation by many workers.

The mass of living and dead vegetation on hillslopes affects the probability of or rate of hillslope mass erosion by increasing the downslope component of mass, thereby increasing the tendency for movement, and increasing the effective force perpendicular to the slope, which increases friction within the soil mass and decreases movement potential. Although these forces affect slope stability in opposite senses, their net effect is generally believed to increase

TABLE 8.2 *Roles of vegetation in regulating hillslope transfer process rates.*

Vegetation component and fraction	Process						
	Solution	Litter-fall	Surface erosion	Creep	Root throw	Debris avalanche	Slump/earthflow
Total biomass							
Loading of slope	0	0	0	+	0	+	-,0,[illegible]
Living vegetation							
Water uptake	-	0	0	--	0	0	--
Nutrient uptake	---	0	0	0	0	0	0
Regulation of snow-melt hydrology	0	0	0	-,0,+	0	++,0,--	+,0,-
Aboveground biomass							
Medium for transfer of wind stress	0	0	0	0,+	+++	++	0,+
Source of litterfall	0	+++	0	0	0	0	0
Roots							
Vertical anchoring	0	0	0	--	---	---	0
Lateral anchoring	0	0	0	--	---	---	-
Living & dead groundcover							
Surface obstruction	0	0	---	0	0	0	0

Note: Vegetation function: increases (+) or decreases (-) transfer process rate.
Significance of vegetation function: questionable or slight (+,-), significant (++,--), substantial (+++,---).

mass movement potential on slopes steeper than 30° (Bishop and Stevens 1964; Swanston 1970; D. H. Gray, pers. comm.). The magnitude of this effect differs as a function of many variables, the principal one being soil depth.

Living vegetation takes up water and nutrients and regulates snow accumulation and melt. Evapotranspiration by plants decreases annual water yield from a watershed (for example, Harr 1976) and shortens the annual period of high soil moisture conditions (Gray 1970). Reduced water yield may decrease solution export from a watershed, and decreased quantities of subsurface water may reduce creep, slump, and earthflow activity. Nutrient uptake by vegetation reduces export of material in solution (for example, Bormann et al. 1969). Regulation of snow hydrology by vegetation may result in either increased or decreased soil moisture peaks during snowmelt events (for example, Anderson 1969; Rothacher and Glazebrook 1968; Harr and McCorison 1979). In the Pacific Northwest rain-on-snow events result in very high soil moisture conditions that trigger debris avalanches (Rothacher and Glazebrook 1968; Day and Megahan 1975) and may cause periods of accelerated creep, slump, and earthflow movement.

Aboveground biomass serves as a medium for transfer of wind stress to the soil mantle. This effect is most conspicuous in the case of blowdown, where uprooted trees may transport both organic and inorganic matter downslope. Blowdown may contribute to the initiation of debris avalanches (Swanston 1969), and Brown and Sheu (1975) have hypothesized that wind stress on the soil mantle may accelerate creep, slump, and earthflow activity. Living and dead organic matter on the ground surface may intercept and temporarily store material moved downslope by surface erosion processes (Mersereau and Dyrness 1972).

Roots may play an important role in stabilizing the soil mantle by vertical and lateral anchoring across potential failure surfaces (for example, Swanston 1970; Nakano 1971). The effectiveness of roots in stabilizing slopes depends on position of the root network relative to potential zones of movement. Roots are most important in stabilizing potential mass failures where failure surfaces are within the rooting zone.

Large organic debris derived from vegetation on adjacent hillslopes controls channel morphology and routing of sediment and water through the stream (Bormann et al. 1969; Swanson et al. 1976; Keller and Swanson 1979). The principal effects of vegetation on channel processes are to physically retard the downchannel transfer of particulate matter, to buttress streambanks, to cause channel deflection, which can increase bank cutting, and to serve as a substrate for biological activity that involves the interchange of dissolved nutrients with stream water (see Chapter 9). Although total sediment yield is largely controlled by input of hillslope processes, the timing of export from a stream may be regulated by large debris in the channel. The presence of debris and temporarily stored sediment may also reduce the rate of channel downcutting, which may, in turn, slow the rate of sediment input by hillslope processes.

Decomposing organic matter and living vegetation may remove certain nutrients from solution in stream water by plant and decomposer organism uptake. There may also be a net input of certain other dissolved nutrients by leaching and decomposition of particulate organic matter. All of this material is eventually exported from a watershed, but whether it is delivered to the gauging site as dissolved or particulate matter may depend on the uptake and dissolution processes.

MATERIAL TRANSFER IN AN OLD-GROWTH FOREST

In order to compare individual and groups of transfer processes, data on transfer rate by each process have been compiled from research results of the coniferous forest biome program, and the Pacific Northwest Forest and Range Experiment Station, USDA Forest Service. This collaborative research has centered on watershed 10 and in the adjacent H. J. Andrews Experimental Forest (see Figure 1.4). This area is located in deeply dissected Tertiary lava flows, dikes, and volcaniclastic rocks in the central western Cascade Mountains (Peck et al. 1964; Swanson and James 1975).

The area of watershed 10 above the sediment basin and gauging flume is 10.2 ha, of which 767 m^2 or about one percent is considered to be stream channel subject to perennial or intermittent surface flow. Gradients of the hillslopes and lower channel average 65 percent and 18 percent, respectively (Figure 8.3). Soils are shallow, only slightly cohesive, and highly permeable.

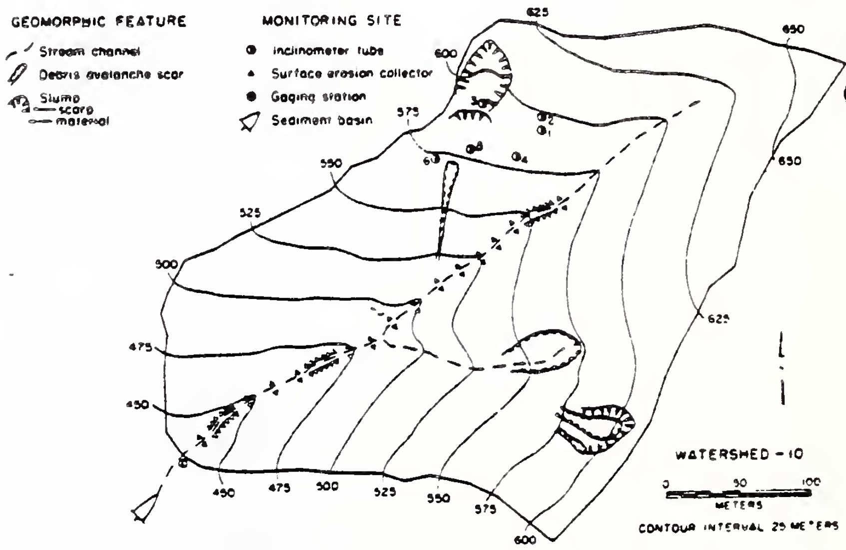

FIGURE 8.3 *Geomorphic features and monitoring sites on watershed 10.*

and they exhibit weakly developed profiles (Harr 1977). Before clearcutting in the summer of 1975, vegetation in the watershed was predominantly *Pseudotsuga menziesii* ranging in age from 400 to 500 years, with younger understory tree canopy composed of *Tsuga heterophylla* on moist sites and *Castanopsis chrysophylla* on dry sites as well as young *Pseudotsuga menziesii.* The dominant old-growth age class appeared to have developed after a disturbance, probably wildfire, about 1475. Portions of the area were again disturbed in about 1800 by a fire that had minor impact on the canopy, but did result in extensive regeneration of *Pseudotsuga menziesii, Tsuga heterophylla,* and *Castanopsis chrysophylla* in the understory. Characteristics of vegetation in the watershed are discussed in detail by Grier and Logan (1977).

The climate of the area is characterized by mild, wet winters and warm, dry summers (Rothacher et al. 1967). Annual precipitation averages between 230 and 250 cm, depending on elevation. At the base of watershed 10 at 440 m elevation more than 90 percent of the annual precipitation falls as rain; snow seldom persists for more than several weeks. Rain-induced snowmelt has resulted in many of the major runoff and hillslope erosion events in the history of the area (Fredriksen 1965).

Many transfer processes are appropriately studied on the spatial scale of watershed 10. These include frequent or continuous processes such as litterfall and surface erosion, as well as less frequent, episodic processes that leave a record of numerous, datable events such as root throw. Processes such as debris avalanches and torrents, which are scattered in time and space, can be viewed better from a wider geographic perspective.

With these constraints in mind, we summarize below available data on transfer of organic and inorganic material by processes operating in a 10-ha, old-growth *Pseudotsuga menziesii* forest. Organic matter is here considered to include all particulate matter made up of carbon (C) compounds plus dissolved organic nitrogen (N) and C. Organic particulate matter includes approximately 1 to 2 percent cations, predominately calcium (Ca), potassium (K), and magnesium (Mg), derived from bedrock and atmospheric inputs. Inorganic matter includes all other material derived from bedrock, volcanic ash fall, and atmospheric sources.

Hillslope Processes

Solution transport of material from hillslopes occurs as water carries dissolved constituents leached from vegetation, soil, and weathering bedrock. There are also atmospheric sources of dissolved mineral material that must be omitted from the total solution export to determine the amount derived from a watershed. Movement of dissolved materials is a pervasive process, operating over the entire watershed and through all strata of the vegetation and soil. Solution transport operates continuously as long as water movement occurs.

but variations exist in response to seasonal and storm event fluctuations in moisture availability, flow-through rate, biological activity, and availability of exchangeable cations and anions. Long-term trends in solution export are controlled by the efficiency of nutrient cycling within an ecosystem and by weathering rate, which is determined by bedrock characteristics, climate, and biological processes.

Input and output of dissolved material have been measured for watershed 10 during water year (WY) 1969 (1 October 1968–30 September 1969) through WY 1973 (Table 8.3). Methods of sample collection and analysis are described by Fredriksen (1972, 1975). Samples were collected at a stream gauging station at the base of the watershed. Since we are attempting to calculate only solution export from hillslopes, it is necessary to assume that the total quantity of dissolved material does not change while water flows from the base of the hillslope through the stream to the flume. Comparison of analyses of

TABLE 8.3 *Input and output of water and dissolved inorganic material (kg/ha) for watershed 10, water years 1969–1973.*

Year	H_2O (cm)	Ortho-P	Na	K	Ca	Mg	SiO_2	Total
1969								
In	253.0	0.01	1.54	0.25	8.77	0.73	—	
Out	169.0	0.42	33.74	1.36	53.62	12.67	—	
Net		-0.41	-32.30	-1.11	-44.85	-11.94	-199.91[a]	-290.42
1970								
In	215.9	0.02	2.62	0.16	2.36	1.53	0.28	
Out	134.6	0.44	25.85	2.26	50.60	12.51	213.57	
Net		-0.42	-23.23	-2.10	-48.24	-10.98	-213.29	-298.26
1971								
In	272.1	0.28	6.64	0.32	3.21	2.78	11.45	
Out	172.3	0.71	37.64	0.98	60.91	17.96	211.43	
Net		-0.43	-31.00	-0.66	-57.70	-15.18	-199.98	-304.95
1972								
In	286.3	0.02	3.98	0.33	4.92	0.82	9.87	
Out	228.52	0.85	43.48	3.25	70.20	16.76	303.75	
Net		-0.83	-39.50	-2.92	-65.28	-15.94	-293.88	-418.35
1973								
In	167.8	0.08	2.59	0.59	0.13	1.07	6.12	
Out	80.16	0.31	13.94	1.46	23.24	6.64	80.01	
Net		-0.23	-11.35	-0.87	-23.11	-5.57	-73.89	-115.02
Average net export		0.46	27.46	1.53	47.84	11.92	196.19	285.40

[a]Not measured. Values assigned from 1971 observations; runoff for 1969 and 1971 were approximately equal.

water samples collected at seeps and at the flume indicates that the concentrations of mineral elements do not change significantly between seep and flume sample sites. Similar analyses for N, however, suggest that there may be 10 to 20 percent uptake because of primary production and decomposition processes in the stream (S. V. Gregory, pers. comm.). Other components of the dissolved organic load do not appear to be significantly altered in the stream environment. Because this is a relatively small change based on preliminary data, we will assume that dissolved organic export measured at the gauging station equals the input to the stream from hillslope areas.

Estimation of dissolved inorganic export is based on analyses for Na (sodium), K, Ca, Mg, and orthophosphate (ortho-P) over the entire five-year period. Analyses for silica (SiO_2) were conducted in WY 1970 through WY 1973 only. Based on these data, annual net transfer is 2.9 t/yr of dissolved inorganic material from hillslope areas.

Dissolved organic matter export was estimated using the following algorithm: based on ten measurements of dissolved organic carbon in water samples collected at the gauging station (S. V. Gregory, pers. comm.), a concentration of 2.0 mg C/liter was applied to total discharge during the initial twenty-four-hour period of fall storms (1 October through 31 December) when flow exceeded 1 cfs (28.3 liters/s). For the remainder of the year a concentration of 0.9 mg C/liter was used. This method was applied to WY 1969 through WY 1973 and the annual estimates were multiplied by two to convert from dissolved carbon to dissolved organic matter. The average of these annual estimates of dissolved organic matter export is 0.3 t/yr, which we assume equals the input to the stream.

Litterfall is the final transfer in a chain of processes involving nutrient uptake by roots, translocation to biomass in the aboveground portion of vegetation, and the fall of litter to the forest floor. Even in predominantly evergreen coniferous forests, litterfall is strongly seasonal, coming mainly in the autumn and early winter months with storm winds, needle and leaf abscission, and snow breakage.

Fall of fine litter into the channel area was directly measured at 0.18 t/yr based on three years of collections in standard 1-m^2 traps along the watershed 10 stream (F. J. Triska, pers. comm.). An estimated 0.15 t/yr of log material is input to the stream, assuming that the standing crop (Froehlich et al. 1972) represents 150 years of input. Total organic matter input to the stream by litterfall is 0.33 t/yr.

Surface erosion of organic and inorganic particulate matter occurs throughout the year in response to wind; diel, storm event, and seasonal variations in moisture and temperature; snow creep, needle ice, and other snow- and ice-driven processes; impact of large pieces of falling litter; and movement of scientists and animals, principally deer, elk, and rodents. Much erosion from steep, bare, mineral soil surfaces results from particle-by-particle movement during dry periods (dry ravel) and by needle ice and the impact of rain and

throughfall drops during wet periods. Mersereau and Dyrness (1972) report that the highest surface erosion rate at a study site in the Oregon Cascades occurred during periods of dry ravel. Areas covered with forest litter often experience movement of surface particles when drying takes place in the spring and early summer. At this time leaves and other litter dry, curl up, and become more susceptible to downslope movement. Overland flow is observed only rarely and very locally in the study area. No evidence of rill formation exists in most forested watersheds in western Oregon except on some debris avalanche scars, road cuts, and other disturbed sites.

Organic and inorganic material moved by surface processes was collected in sixty-four fifty-cm-long erosion boxes along the perimeter of the watershed 10 stream. The rate of surface movement into the total length of boxes may be extrapolated to the entire perimeter of the stream to estimate total annual surface movement transfer from hillslopes to the stream. Based on two years of observations, 0.30 t of organic material and 0.53 t of inorganic matter were transferred annually into the stream channel.

Soil creep is the slow deformation within and between individual soil particles in response to gravitational stress but without the development of discrete failure planes. Creep grades into more rapid mass movement processes that do have well-developed failure planes, however, and distinction among these processes is somewhat arbitrary (Terzaghi 1950).

Soil creep is difficult to monitor, because it affects virtually all sloping soil masses and movement is slow, generally less than 1 cm/yr. Although precise measurements are required, it is difficult to establish stable reference points. Creep measurement in watershed 10 and at nearby sites have been made with a set of inclinometer installations (D. H. Gray, pers. comm.).

Annual measurement with an inclinometer revealed downslope deflection occurring at soil depths (measured vertically) of up to 4 m. The net downslope deflection was used to calculate an average creep rate for each inclinometer tube (Table 8.4). Displacement ranged from 0.25 to 0.46 mm for four of the tubes, but the other two tubes, which were located on a bench, experienced no significant movement. Because this landform is of limited areal extent (Figure 8.3), creep rates observed for tubes on steeper sites are considered more typical of the watershed. The variability and maximum values of creep rates observed in watershed 10 are comparable to creep rates measured in similar topographic and geologic settings (D. H. Gray, pers. comm.; D. N. Swanston, pers. comm.).

There are certain limitations on the usefulness of the data in Table 8.4. Using a cumulative five-year record of creep observations to calculate annual movement minimizes the analytical problem that at certain depths and in certain tubes annual movement is less than the resolution of the instrument, approximately 0.6 mm (D. N. Swanston, pers. comm.). An additional consideration in evaluating these data is that the bases of the inclinometer tubes may not be fixed to stable bedrock. Therefore additional movement may be taking place

TABLE 8.4 *Creep measurements with inclinometer installation.*[a]

Tube number	Hillslope angle (degrees)	Depth (m)	Annual surface displacement (mm/yr)	Average displacement over entire depth of tube (mm/yr)	Period of record
1	32	3.96	0.60	0.25	1969–1974
2	35	2.44	0.93	0.46	1969–1974
3	27	4.27	0.17	<0.01	1969–1974
4	34	3.96	1.57	0.32	1969–1974
5	27	3.66	0.34	0.04	1969–1973
6	41	3.96	0.73	0.40	1969–1974

[a]D.H. Gray, personal communication.

that is not recorded by these installations. Despite these limitations, we make a minimum estimate of creep transfer of material to the channel by assuming that a block of soil 2.65 m thick, the average thickness for the watershed (R. D. Harr, pers. comm.), crosses the 1150 m of channel perimeter at a rate of 0.35 mm/yr, the depth-integrated average creep rate of tubes 1, 2, 4, and 6. Based on these assumptions and an average bulk density of 1.0 t/m^3 (R. L. Fredriksen, pers. comm.), creep supplies 1.1 t/yr of inorganic material to the channel.

Organic matter transfer by creep may be estimated by creep rate (0.35 mm/yr) × channel perimeter (1150 m) × cosine of slope angle (33°) × biomass per m^2 of watershed (0.125 t/m^2; Grier and Logan 1977). Estimated organic matter transfer to the channel by creep is 0.04 t/yr.

Root throw by living and recently dead trees downed by strong winds generally move organic and inorganic material downslope. When root systems undergo extensive decay before a tree falls, binding between soil and roots is lost and little soil is moved. Pit and mound microrelief due to root throw has been well described in the gentle topography of the eastern United States, where root throw is an important soil disturbance factor (for example, Denny and Goodlett 1956; Stone 1975). In stands of large, old-growth forests on steep slopes erosion by root throw is accentuated by massive root wads and their tendency to slide downslope. Such events are episodic, occurring in windstorms that have a return period of several years to decades. Deep-seated earth movement by slump, earthflow, and creep processes can lead to tipping and even splitting of trees, thereby increasing their susceptibility to blowdown.

Root throw is such a sporadic phenomenon that a long-term historical record is necessary to validly estimate its occurrence. Fortunately, root wads, soil mounds, and pits from which roots and soil were removed are clearly recognizable and their features are datable for more than a century following the event. Events are dated by counting rings of trees growing on the pit, soil

mound, or the downed tree. The persistence of pits and mounds attests to slow rates of surface erosion and creep.

In watershed 10, 112 mapped root-throw sites have abundant root and bole material still present. Dendrochronologic observations and the stage of decay of residual organic matter (P. Sollins, pers. comm.) suggest that the inventoried root-throw occurred to large, old-growth trees in the past 150 years. Root-throw sites are distributed rather uniformly over most of the watershed with some concentration along the lower slopes, particularly along the north side of the stream. Direction of fall was predominantly downslope.

The annual quantity of sediment supplied to the stream is difficult to determine, because most root wads in the stream slid >10 m down to the channel, loosening soil along the way. Eight of the inventoried root wads reached the stream. If we assume that these events occurred over the past 150 years, that each one transported or pushed 2 m^3 of soil to the stream, and that soil bulk density was 1.0 t/m^3, annual transfer to the channel would be 0.1 t of inorganic material. The organic matter in roots of <5 cm diameter in the root wad of a 120-cm-dbh old-growth Douglas-fir is estimated to be about 2 t (Santantonio et al. 1977; D. Santantonio pers. comm.), which would result in 0.1 t/yr of annual organic matter transfer to the channel.

Debris avalanche is used here as a general term for rapid, shallow soil mass movements, including events that have been classed by other workers as debris flows, slides, and rapid earthflows (Varnes 1958). These mass movements commonly occur as a result of periods of intense rain or rain plus snowmelt while the soil is already very moist (Fredriksen 1965; Dyrness 1967). Occurrence of debris avalanches are scattered in both time and space. In the H. J. Andrews Experimental Forest, for example, it has taken storms of about a seven-year return period to trigger debris avalanches in forested areas (F. J. Swanson, pers. comm.).

Debris avalanches are commonly believed to be a dominant erosion process in landscapes such as watershed 10 and similar terrains in the H. J. Andrews Experimental Forest (Fredriksen et al. 1975; Swanson and Dyrness 1975); however no debris avalanches of larger than 75 m^3 occurred in the watershed during at least the past century. Their earlier occurrence is suggested by landforms interpretable as avalanche scars (Figure 8.3). The oldest trees growing on these features range in age from 200 to more than 400 years.

Because a 10-ha watershed offers a limited record of its debris avalanche history, examination of a larger area of similar terrain that would contain more recent, datable events for study is useful. A record of all debris avalanches greater than 75 m^3 has been compiled for the period from 1950 to 1975 in the H. J. Andrews Experimental Forest (Dyrness 1967; Swanson and Dyrness 1975). In this twenty-six-year period fourteen debris avalanches occurred in the 20-km^2 portion of the forest that is similar to watershed 10 in terms of soil, topography, and forest cover. The annual frequency of debris avalanches was 0.27 $event \cdot km^{-2} \cdot yr^{-1}$, or 0.0027 event/yr in 10 ha, the area of watershed 10.

The return period for a single event, the inverse of event frequency, is 370 years in a 10-ha area, assuming equal probability of occurrence over the inventoried area.

The fourteen debris avalanches in the terrain similar to watershed 10 transported a total of 30,870 m^3 of soil, an average of 2205 m^3 per event (for a description of field methods see Swanson and Dyrness 1975). The dimensions of mapped debris avalanche scars in watershed 10 indicate that this is a reasonable estimate of the volume of recent, significant (greater than 75 m^3) debris avalanches in the watershed.

Twelve inventoried events entered streams and 99 percent of the volume of material transported was readily available to perennial or intermittent streams. The straight, steep slopes of watershed 10 present no impediments to debris avalanches on their way to the stream. Assuming that all of the debris avalanches move soil from hillslopes to stream channels, and that soil bulk density is 1.0 t/m^3 (R. L. Fredriksen, pers. comm.), transfer of inorganic particulate matter occurs at a rate of 6.0 t/yr.

Debris avalanches also transport organic matter to the stream. The average plan view area of the fourteen inventoried debris avalanches is about 1200 m^2, calculated from average volume, assuming average soil depth of 1.5 m and a slope of 36°. Assuming 0.125 t/m^2 of terrestrial biomass (Grier and Logan 1977), such a debris avalanche would transport 150 t of biomass. If one average-sized event occurred in the watershed in 370 years annual biomass transfer would be 0.41 t/yr.

Slump and earthflow landforms are developed where masses of earth undergo rotational or translational movement along discrete failure planes or zones of failure (Varnes 1958). In the western Cascades these features range in size from less than 1 ha to hundreds of hectares (Swanson and James 1975). Increased rates of slump and earthflow may occur in response to periods of heavy precipitation (R. D. Harr, pers. comm.; F. J. Swanson pers. comm.). In other instances, slump and earthflow features appear to have been inactive for hundreds or thousands of years. Where slumps and earthflows encroach on streams, the channel cross-section is progressively constricted and banks are oversteepened (Swanson and Swanston 1977).

Rotational and translational slump and earthflow movement has produced landforms covering approximately 6 percent of the area of watershed 10 (Figure 8.3). Field observations suggest that the slumps have been dormant for decades or perhaps centuries. Trees growing on the slump benches and deposits exhibit no signs of having experienced splitting, tilting, or periods of eccentric growth typical of vegetation growing on moving ground (Swanson and Swanston 1977). In areas of the Andrews Experimental Forest where differential movement is occurring at rates greater than 1 cm/yr, open tension and shear cracks are formed. Since none of these features has been observed in the slump areas of watershed 10, we conclude that during the past century slump movement in watershed 10 has been negligible.

Stream Channel Processes

Solution transport is a persistent process, operating under all streamflow conditions. Dissolved material in stream water is derived from the dissolved load of groundwater, direct atmospheric inputs, throughfall, and leaching of particulate matter weathering or decomposing in the channel. Values of dissolved inorganic and organic material determined for watershed 10, were 2.9 and 0.3 t/hr, respectively.

Suspended sediment is made available to the stream by all hillslope processes that transport particulate matter and by lateral and vertical cutting of the streambanks and bed. Small amounts of suspended sediment are carried by streams throughout the year, but most of the total annual load is transported during a few large storms. During stormflow fine particulate matter is scoured from streambed and bank deposits of alluvium and colluvium; it is produced by the breakdown of larger particles, and released from temporary storage in coarse alluvium when bedload movement commences.

The input and output of fine particulate matter at watershed 10 have been measured over a five-year period by Fredriksen (1975). Inputs come primarily as atmospheric fallout of particles greater than 0.05 mm in size. Particulate matter was filtered from precipitation collected in birdproof precipitation collectors. The relative contributions of natural and man-influenced and distant and local sources of atmospheric particulate matter are not known (Fredriksen 1975). Aeolian entrainment of organic particulates has not been quantified in ecosystem studies, although several mechanisms for entrainment have been suggested (Fish 1972).

Suspended sediment export was sampled at the flume with a pumping proportional water sampler (Fredriksen 1969, 1975). Both the suspended sediment and atmospheric input samples for WY 1972 and WY 1973 were analyzed for carbon (Fredriksen 1975) and the total organic component was calculated assuming that it is twice the amount of carbon.

Because of the ambiguous status of fine particulate inputs to the watershed, we report export of suspended sediment both with and without subtracting apparent atmospheric inputs. Suspended organic matter export from the watershed was 0.12 t/yr, or 0.07 t/yr when apparent atmospheric input is subtracted. Suspended inorganic matter export amounted to a gross value of 0.78 t/yr, and 0.56 t/yr when atmospheric dust input is subtracted.

For purposes of comparison, we have also examined data for gross organic plus inorganic suspended sediment export from four nearby experimental watersheds with soil, forest cover, and geomorphic conditions similar to watershed 10. The average annual fine particulate export for these watersheds ($98\ kg \cdot ha^{-1} \cdot yr^{-1}$) is in close agreement with the five-year average for watershed 10 ($90\ kg \cdot ha^{-1} \cdot yr^{-1}$).

Bedload transport commonly occurs during only a few major runoff events each year. Material available for bedload transport is temporarily stored in the

stream channel, commonly behind large pieces of organic debris, boulders, or outcrops of resistant bedrock. Concentrations of large organic debris in a channel greatly influence its sediment storage capacity. Therefore, on a time scale of years or decades, bedload transport is regulated by changes in channel storage capacity, rates of sediment supply from hillslopes, and hydraulic forces tending to move sediment out of the channel.

Bedload export from watershed 10 was monitored for only two years before logging. The record is inadequate to characterize this erosion process for the basin, so we use the longer record available for experimental watersheds with generally similar forest cover, topography, soil, and geomorphic conditions.

A total of twenty-nine watershed years of bedload data is available for watersheds 1, 2, 3, 9, and 10 in and adjacent to the H. J. Andrews Experimental Forest (Table 8.5; Fredriksen 1970, pers. comm.).

Bedload from watershed 10 was collected in large nets placed at the downstream end of the flume. Sampling was done on a continuous basis during low flow using a 80 μm net and periodically during high flow using a 1 mm net. Bedload export from other watersheds was measured by annual surveys of settling basins at the bottom of each watershed. In all cases reported values are minimum estimates of the true value, because trapping efficiencies of the basins are probably less than 100 percent particularly during peak periods of bedload transport.

Bedload export for all watersheds has been 93 $kg \cdot ha^{-1} \cdot yr^{-1}$. For the four watershed years at watersheds 9 and 10, 65 percent of total bedload was com-

TABLE 8.5 *Average annual bedload and suspended load export from forested watersheds in the H. J. Andrews Experimental Forest.*[a]

Watershed	Period of record (water year)	(yr)	Bedload ($kg \cdot ha^{-1} \cdot yr^{-1}$)	Suspended load ($kg \cdot ha^{-1} \cdot yr^{-1}$)
1	1957–1962	6	61	78[b]
2	1957–1972	16	122	—
	1957–1967	11	—	134[b,c]
3	1957–1959	3	95	130[b]
9	1974–1975	2	21	—
	1969–1973	5	—	31[d]
10	1973–1974	2	32	—
	1969–1973	5	—	90[d]

[a]Bedload average for 29 watershed years = 93 $kg \cdot ha^{-1} \cdot yr^{-1}$. Suspended sediment average for 30 watershed years = 98 $kg \cdot ha^{-1} \cdot yr^{-1}$.

[b]Based on depth integrated hand sample at flume.

[c]Includes two highest floods on record in December 1964 and January 1965.

[d]Based on integrated samples collected with a pumping proportional sampler.

posed of inorganic material. Therefore the estimated inorganic matter export from watershed 10 is 0.60 t/yr, and coarse particulate organic matter export is 0.33 t/yr.

Debris torrents are commonly triggered by debris avalanches entering the channel from adjacent hillslopes (Swanson et al. 1976). A debris avalanche may maintain its momentum, becoming a debris torrent as it moves directly down the channel, scouring the streambanks and bed. Debris torrents are also initiated by mobilization of debris that had previously entered the channel by a variety of processes such as debris avalanches, windthrow, and bank cutting. Debris torrents are infrequent events, occurring in response to major storms. Most torrents in small streams are in part regulated by the debris avalanche potential of hillslopes in the basin. Most small basins experience a torrent less frequently than once in a century.

As in the case of debris avalanches, debris torrent history is not adequately represented in a small area the size of watershed 10. This analysis is based on debris torrent activity in the H. J. Andrews Experimental Forest from 1950 through 1975. Nine debris torrents occurred in 20 km^2 of terrain with geomorphic and forest conditions similar to those of watershed 10. It follows that there were 0.017 debris torrents $\cdot km^{-2} \cdot yr^{-1}$, which is an annual probability of 0.0017 that an event will occur in an area the size of watershed 10, or an estimated average return period of one event in 580 years.

Eight of the nine inventoried events were triggered by debris avalanches. Debris avalanches also appear to be the dominant triggering mechanism of debris torrents in watershed 10 where steep, smooth slopes lead directly from debris avalanche-prone areas to the stream channel (Figure 8.3). The average length of inventoried events was 370 m. In watershed 10 the two most prominent debris avalanche sites would initiate torrents at points about 260 m upstream from the flume.

If the entire mass of an average debris avalanche (2205 t inorganic matter and 150 t organic) were to move through the channel and past the flume as a debris torrent, it would involve 5.3 t/yr export of inorganic matter and 0.26 t/yr export of organic material, based on the occurrence of a single event during the estimated 580-year return period. This hypothetical debris torrent would also entrain alluvium, organic debris, and soil from the channel and adjacent banks. Of course the volume of material in a channel varies greatly with recent history of storms, vegetation, and geomorphic processes. The prelogging concentrations of organic and inorganic material in the channel of watershed 10 represented a moderate level of channel loading relative to observations made in other streams (Froehlich et al. 1972; Froehlich 1973). Before logging, approximately 40 t of organic matter was in a four-meter-wide strip along the 260 m of the channel between the most likely point of introduction and the flume (Froehlich et al. 1972). Inorganic matter in this strip along the channel is estimated to be 470 t, assuming a 0.3 m average depth of alluvium and soil with bulk density of 1.0 t/m^3 and a six-meter-wide torrent track. If all of this

material were entrained by a debris torrent, total debris torrent export from the watershed (entrained material plus debris avalanche material) would be 4.6 t/yr of inorganic matter and 0.33 t/yr organic matter.

Accuracy and Limitations of Estimates

The usefulness of erosion process rate estimates summarized in Table 8.6 is directly related to their accuracy. Unfortunately, in most cases it is impossible to calculate an estimate of accuracy objectively owing to lack of data to quantify some of the key assumptions used to make the erosion rate estimate. Accuracy of estimates is a particular problem in the cases of debris avalanches, torrents, and bedload transport, which occur mainly during infrequent, large storms. A few storms can dominate even a record of more than 25 years such as that available for mass movement processes. This period of record contained only three of the top ten annual peak flows recorded in the fifty-nine-year record for the McKenzie River at McKenzie Bridge, the nearest long-term gauging site (Dalrymple 1965). The highest peak flow of the entire fifty-nine-year record, however, occurred in December 1964, and on Lookout Creek in the H. J. Andrews Forest it was 27 percent higher than the estimated fifty-year recurrence interval event (Waananen et al. 1971). Because the return period of this event was much greater than the length of record, these rates of debris avalanches, torrents, and bedload transport may be overestimated. The available record is too brief to characterize these episodic processes accurately. Furthermore, transfer rates for the mass-movement processes are based on small sample sizes. Therefore we estimate that the measures of debris avalanche and debris torrent rates have an accuracy of no better than +60 percent to −100 percent. As a result of more frequent occurrence of bedload transport and sampling inefficiencies, we estimate accuracy of bedload transfer rate to be ±60 percent.

The more frequent or continuous processes are better characterized by the available data. Rates of solution transport and to some extent creep and suspended sediment are related to total annual water yield. Analysis of long-term streamflow records suggests that the transfer estimates for solution transport, creep, and suspended sediment are good representations of the past twenty-five years, but may overestimate rates for the past sixty-five years, which includes some drier periods earlier this century. Estimated accuracy of transfer rates for these processes is about ±30 percent. For all other process rates, we estimate an accuracy of approximately ±50 percent.

It is important to note that these estimates of transfer rates apply to old-growth forest conditions; some process rates vary significantly for different stand conditions on the same landscape. This is particularly true immediately following ecosystem disturbances such as wildfire or clearcutting. These factors are considered further in a later section.

EROSION UNDER FORESTED CONDITIONS

Material transfer data may be examined in terms of total watershed-ecosystem export, contrast of total hillslope and channel transfer, and comparisons among processes. In each case we compare the roles of episodic and the more frequent or continuous processes, the relative importance of organic versus inorganic transfer, and dissolved versus particulate material export.

Watershed-Ecosystem Export

Total watershed-ecosystem export by channel processes is 0.99 $t \cdot ha^{-1} \cdot yr^{-1}$ (Table 8.6). This figure includes debris torrents, which account for an estimated 50 percent of the total, even though it is assumed that only one event occurs in 580 years. Total export by the commonly measured processes of dissolved, suspended, and bedload sediment transport is 0.5 $t \cdot ha^{-1} \cdot yr^{-1}$. Par-

TABLE 8.6 *Transfer of organic and inorganic material to the channel by hillslope processes (t/yr) and export from the channel by channel processes (t/yr) for watershed 10.*

Process	Inorganic matter	Organic matter
Hillslope processes		
Solution transfer	3	0.3
Litterfall	0	0.3
Surface erosion	0.5	0.3
Creep	1.1	0.04
Root throw	0.1	0.1
Debris avalanche	6	0.4
Slump/earthflow	0	0
Total	10.7	1.4
Total particulate		
Including debris avalanche	7.7	1.1
Excluding debris avalanche	1.7	0.7
Channel processes		
Solution transfer	3.0	0.3
Gross suspended sediment	0.8	0.1
Net suspended sediment	0.6	0.1
Bedload	0.6	0.3
Debris torrent	4.6	0.3
Total	8.9	1.0
Total particulate		
Including debris torrent	5.9	0.7
Excluding debris torrent	1.3	0.4

ticulate matter composes 34 percent of this total. Organic matter makes up 24 percent of the particulate matter export (excluding debris torrents) and 9 percent of total dissolved export.

Values for organic and inorganic, particulate and dissolved materials are summarized in Table 8.7 along with similar data from Hubbard Brook, New Hampshire, the only site for which complete comparable data exist. Characteristics of these two watershed ecosystems (Table 8.8), are the basis for contrasting the two systems.

In the cases of all four constituents, export values for watershed 10 (even excluding debris torrents) exceed those for watershed 6, Hubbard Brook, by a factor of at least 2. Some of the apparent contrasts may arise from differences in methods and efficiencies of sample collection and analysis, from possible differences in the relative magnitude of major storms (in terms of return period) that occurred within the respective sampling periods, and possibly from differences in magnitudes of storms of comparable return period. Contrasts in estimated export from the two watersheds is so great for each of the variety of forms of export, however, that much of the difference may be due to real differences in system behavior.

Systems may differ in terms of the availability of material to be transported and the energy available to transport material. Export of dissolved constituents

TABLE 8.7 *Export of dissolved and particulate organic and inorganic material ($kg \cdot ha^{-1} \cdot yr^{-1}$) from watershed 10, H. J. Andrews Experimental Forest, Oregon, and watershed 6, Hubbard Brook, New Hampshire.*[a]

	Watershed 10 H. J. Andrews Experimental Forest		Watershed 6 Hubbard Brook
	Excluding debris torrents	Including debris torrents	
Dissolved			
Organic	30	30	15
Inorganic	300	300	58
Total	330	330	73
Particulate			
Organic	40	70	10
Inorganic	130	590	15
Total	170	660	25
Total			
Organic	70	100	25
Inorganic	430	890	73
Total	500	990	98

[a]Bormann et al. 1974.

tends to be directly related to total annual discharge, while particulate matter export is controlled by high discharge events (Bormann et al. 1974). Availability of particulate material for transport from a channel depends on the input rate of material from adjacent hillslopes and stream power, a function of discharge and channel gradient. Availability of material for transport in solution is determined in part by the ability of the biota to immobilize nutrients in living vegetation and detritus. In the case of inorganic material, rates of weathering and decomposition also determine availability of dissolved material.

Dissolved organic matter export from watershed 10 is twice that of watershed 6, which may largely reflect the higher annual runoff from the Oregon site (Table 8.8). The five times greater standing crop of organic matter in watershed 10 also results in more material available for leaching from the ecosystem. The contrast is especially striking considering that dead biomass in watershed 10 equals total biomass reported for the Hubbard Brook site.

The condition of greater biomass available for export from watershed 10 doubtless contributes to its fourfold higher particulate organic matter export. Availability of material for export from watershed 10 is also affected by the much steeper hillslopes, which result in greater rate of particulate matter transfer to the channel. The much higher litterfall rate in watershed 10 also contributes to greater particulate organic matter export. These factors must more than compensate for slightly higher gradient in the lower 100 m of channel and

TABLE 8.8 *Characteristics of watershed 10, H. J. Andrews Experimental Forest, Oregon, and watershed 6, Hubbard Brook, N.H.*

		Slope		Runoff					
Watershed no.	Area (ha)	Total watershed (%)	Channel lower 100 m (%)	Av ($l/sec^{-1}/ha^{-1}$)	Peak ($l/sec^{-1}/ha^{-1}$)	Av annual peak ($l/sec^{-1}/ha^{-1}$)	Av annual (cm)	Av annual air temp (°C)	Dominant tree species
10	10.2	60	18	0.5	19	12	156	9.5[a]	*Pseudotsuga menziesii, Tsuga heterophylla*
6	13.2[d]	26[c]	21[c]	0.3[c]	25[c]	15[c]	80[c]	4.3[f]	*Acer saccarum, Fagus grandifolia, Betula alleghaniensis*[d]

[a]Waring et al. 1978.
[b]Grier and Logan 1977.
[c]P. Sollins, personal communication.
[d]Bormann et al. 1970.

higher average annual peak discharge for the five-year periods of record at the Hubbard Brook site.

Particulate organic matter export from small watersheds may also be regulated by retentiveness or roughness of the channel system. Boulders and living and dead vegetation in and adjacent to the channel slow downstream routing of particulate matter, providing more opportunity for biological processing and export from the system by respiration and leaching. In streams at Hubbard Brook boulders are the dominant elements of bed roughness, whereas large woody debris is the major controller of particulate matter routing through a small Oregon stream. Both systems appear to have high roughness and therefore a tendency to retain organic detritus until it is processed by aquatic organisms (Bormann et al. 1969; Sedell and Triska 1977).

The export of dissolved inorganic matter from watershed 10 exceeds that of watershed 6 by about sixfold (Table 8.7). Part of this difference is accounted for by higher total runoff from watershed 10, but a more important factor is the greater weathering rate of soils and bedrock at the Oregon site. The higher weathering rate at the Oregon site is due to higher temperatures and precipitation and the mineralogy of altered volcanic rocks. Hydrothermal alteration of these volcanic breccias resulted in formation of readily weathered secondary minerals and amorphous materials even before the rocks were subjected to the modern weathering environment. Weathering of bedrock and the compact till that blankets the Hubbard Brook watershed proceeds at a slower pace in response to mineralogic properties and the weathering environment of the soil.

Age (yr)	Biomass (watershed) (kg/m^2) Live	Biomass (watershed) (kg/m^2) Dead	Litterfall incl. stems (kg/m^2)	Bedrock	Soil
100 to 150 and 400 to 500	86[a]	38[a]	1.1[a,c]	volcanic breccias, tuffaceous sediments propylitically altered	Dystrochrept, poor horizon development
55[e]	16[e]	22[e]	0.57[g]	Quartz-biotite gneiss, sillimanite-zone metamorphism[f]	Spodosols (Haplorthods), moderate profile development[f]

[e]Bormann et al. 1974.
[f]Federer 1973.
[g]Gosz et al. 1976.

Differences in ability of the biota to immobilize cations and thereby regulate dissolved inorganic matter export from the two watersheds are probably minor, because 70 to 80 percent of this export component is made up of SiO_2 and Na, which are not significantly accumulated in the plant or microbial biomass of these ecosystems.

Estimated particulate inorganic matter export from watershed 10, excluding debris torrents, is about nine times greater than that of watershed 6 (Table 8.7). As described in the case of particulate organic matter, in their lower reaches the two channel systems appear to have similar transport capability, except in the case of debris torrents. Therefore, marked differences in particulate inorganic matter export probably arise from a more rapid rate of sediment input to the channel from steeper hillslopes of watershed 10.

Comparison of Hillslope and Channel Material Transfer

Estimated total particulate organic and inorganic inputs to the watershed 10 channel are greater than comparable output values. Much of the input/output difference for particulate organic matter is due to biological utilization (see Tables 10.3 and 10.4), whereas the difference for particulate inorganic matter is within the error of input and output estimations. If input generally exceeds output, the streambed should experience net aggradation. Actually the watershed 10 stream is at the bottom of a steep-sided, V-notch valley, indicating a long history of downcutting. Short-term watershed budget studies and examination of sediment routing function and history of large woody debris in streams (Swanson et al. 1976) suggest, however, that channel systems may be sites of net increase in storage for long periods of time interrupted by infrequent, major flushing events. Consequently, such forested streams may be aggrading on the time scale of years and decades, while experiencing degradation on a broader time scale.

Comparison of Processes

The material transfer data may also be evaluated in terms of relative roles of various processes. Rates of processes accounting for inorganic matter transfer vary over a broad range (Table 8.6). The most infrequently occurring processes, debris avalanches and torrents, appear to be dominant, although only one event occurs every few centuries on the average. Solution transfer, one of the most continuous processes, is the second most important mechanism of inorganic matter transport. Processes of secondary importance include creep, surface erosion, and root throw on hillslopes and suspended sediment and bedload transport in the channel. Litterfall, slump, and earthflow processes are presently insignificant in terms of transporting inorganic matter.

In the case of organic matter transport there is much less variation in the relative importance of most processes (Table 8.7). Among hillslope processes debris avalanche, surface erosion, litterfall, solution, and root throw each supplies material to the channel at rates of about 0.1 to 0.4 t/yr. Particulate organic matter transfer by creep is about an order of magnitude lower, and slump and earthflow processes are presently negligible. Estimated organic matter export rate for each channel process is in the range of 0.1 to 0.3 t/hr. Episodic processes are relatively less important than more continuous ones in transporting organic matter.

EFFECTS OF ECOSYSTEM DISTURBANCE

As a result of numerous interactions between vegetation and material transfer processes in forests, severe disturbances of vegetation affect transfer processes throughout forested watershed ecosystems. This fact has been amply demonstrated in terms of sediment yield from paired forested and manipulated watersheds in areas of diverse climate, vegetation, and geomorphic setting (for example, Fredriksen 1970; Brown and Krygier 1971; Bormann et al. 1974; Fredriksen et al. 1975). Results of watershed manipulation experiments in the Pacific Northwest have been highly varied, depending on treatment, terrain, and history of past disturbances. Effects of timber harvest on sediment yield range from negligible in the case of two watersheds of low slope (7 to 12 percent) that were 25 percent clearcut (Fredriksen et al. 1975) to a twenty-three-fold increase in suspended sediment export over a fourteen-year period from watershed 3 in the H. J. Andrews Experimental Forest, which was 25 percent clearcut and 6 percent roaded (Fredriksen 1970; pers. comm.). Watershed 10 was clearcut and cable yarded in summer 1975 and early stages of postlogging erosion are being examined.

Initial observations in watershed 10 and other experimental watersheds suggest that postclearcut watershed export comes from three sources, each associated with a specific time frame: (1) material input to the channel during falling and yarding operations, consisting of mainly fine, green organic matter and some mineral soil; (2) material that had entered the channel by natural processes and was in temporary storage behind debris obstructions before logging, but is released from storage when large pieces of organic debris are removed from the channel during logging; and (3) material input to the channel by hillslope erosion processes following logging. A general phasing of watershed export of materials from these three sources may occur with material from source 1 mainly leaving the watershed in the first one to three years following cutting, source 2 gaining importance in the latter part of this period, and the postlogging hillslope erosion (source 3) becoming a dominant source several years after cutting. This phasing or routing of material through a watershed is an important element of ecosystem response to disturbance and it is relevant to

interpretation of sediment yield data. Sediment yield from manipulated watersheds is commonly interpreted in terms of hillslope transfer processes, where in some cases it may result from changes in channel storage.

Ultimately, postcutting studies in watershed 10 will test hypotheses concerning the role of revegetation in returning individual process rates to levels characteristic of forested conditions. Each process has a different magnitude and timing of response to deforestation due to differences in interactions between transfer processes and vegetation. Hypothetical trajectories of several hillslope process rates following cutting are shown in Figure 8.4. The timing of change in debris avalanche potential is partly a response to the timing of decay of root systems from the precutting vegetation and the buildup of root systems in the postcutting stand. The net effect of this and possibly other factors in areas of the H. J. Andrews Experimental Forest similar to watershed 10 has been a 2.8 times increase in debris avalanche erosion over about a twelve-year period following clearcutting (Swanson and Dyrness 1975). Surface erosion involves a pulse of material transfer during and soon after the logging operation followed by a period of recovery. Timing of recovery is controlled by the rate of reestablishment of ground cover or development of a residual armor layer of coarse soil particles. Soil solution transport is regulated by nutrient and water uptake by vegetation. With recovery of leaf area and rates of primary production, a proportion of available nutrients is incorporated into biomass and a smaller amount is flushed from the system. Additionally, recovery of vegeta-

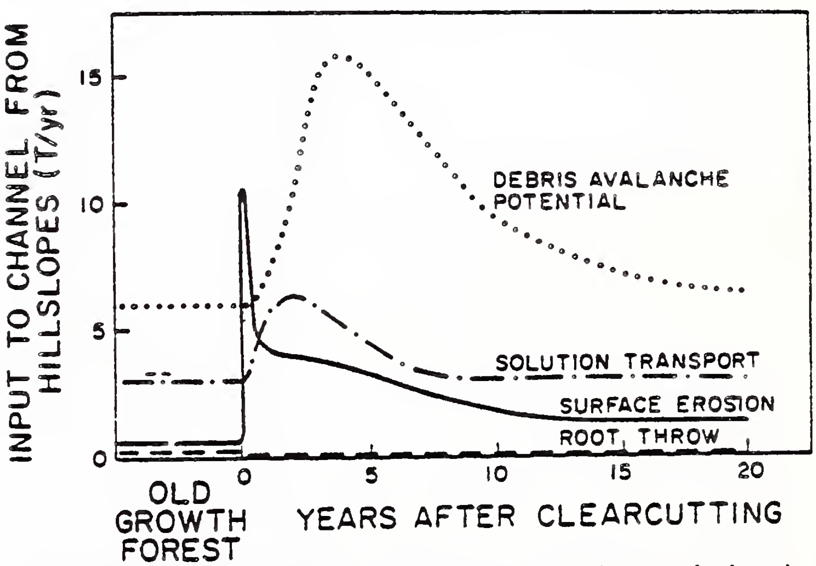

FIGURE 8.4 *Hypothetical trajectories of potential rates of selected hillslope transfer processes after clearcutting of watershed 10.*

tion reduces annual water yield to predeforestation levels over a period of a decade or more. Root throw within the deforested watershed is eliminated as a significant process for several decades until regeneration trees are large enough to be subject to blowdown. Rates of other hillslope and channel processes and channel storage conditions may all vary somewhat out of phase with one another, although there is some degree of interdependence since hillslope processes supply material for transport by channel processes.

To fully assess effects of ecosystem disturbance on material transfer, a broad historical perspective is needed. Typically in the assessment of management impacts, manipulated systems are compared with forested reference or benchmark watersheds; however, most natural, unmanaged watersheds are subject to periodic severe disturbance. Consequently material transfer history under both managed and natural conditions is composed of periods with transfer rates characteristic of established forest conditions interspersed with periods of accelerated transfer spanning up to several decades following severe disturbance of the ecosystem.

In many Pacific Northwest *Pseudotsuga menziesii* forests, natural premanagement disturbances during the past 1000 years have been predominantly major crown fires with a return period of several centuries. Erosional consequences of this type of disturbance are doubtless great, but unknown in steep landscapes. Timber harvest in this area is expected to recur at 80 to 100-year intervals, and its consequence in terms of material transfer is understood in only a preliminary fashion. Based on these assumptions and data, we construct a hypothetical variation in sediment yield from watershed 10 relative to a 500-year history of wildfire and a projected pattern of future management activities and related accelerated material transfer (Figure 8.5). Clear understanding of timber management impact in such a long-term perspective will require knowledge of frequency and consequences of both management and natural premanagement disturbances of the ecosystem.

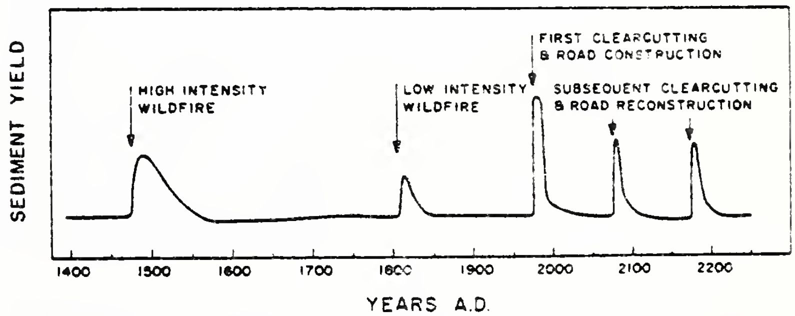

FIGURE 8.5 *Hypothetical history of sediment yield in response to vegetation disturbances on watershed 10.*

SUMMARY

Physical transfer of organic and inorganic matter is an important part of ecosystem behavior. At the system level, physical processes of material transfer account for principal nutrient cycling fluxes. From the standpoint of vegetation distribution, erosion and deposition create constrasting habitat opportunities for aquatic and terrestrial organisms. Erosion also reduces the nutrient capital of a site and affects the course of succession.

Material transfer in steep, forested watersheds of the coniferous forest biome is accomplished by processes that interact with one another and with various components of vegetation. The principal hillslope processes are solution transfer, litterfall, surface erosion, debris avalanche, creep, root throw, slump, and earthflow. These processes supply organic and inorganic material to the channel where downstream transport then occurs as dissolved and suspended material, bedloads, and debris torrents.

These processes operate on a variety of scales in time and space. At one extreme, debris avalanches and torrents may occur in a small watershed only once every few centuries under forested conditions and an event affects only a small percentage of the landscape. On the other hand, creep, litterfall, and solution transfer operate continually over the entire watershed.

These processes are highly interactive. Some events may directly trigger other processes, as in the case of root throw, which may instantaneously initiate a debris avalanche. One process may also set the stage for the occurrence or acceleration of another process, such as the baring of mineral soil by root throw and debris avalanche, which leads to a period of increased surface erosion. Processes also supply material for transport by other processes, so that transfer of a particular particle of soil through a watershed occurs as a series of steps in a variety of modes of transport.

Vegetation increases the rates of some transfer processes while decreasing others. Rooting strength, the mass of vegetation on a hillslope, and hydrologic effects of vegetation regulate rates of debris avalanche, creep, slump, and earthflow activity. Root throw occurs because standing trees serve as a medium for transfer of wind stress to the soil mantle. Nutrient uptake by plants and other processes regulate export of dissolved material. Litterfall results in a net downslope transfer of particulate organic matter as a result of nutrient uptake, incorporation into biomass, and subsequent abscission or pruning by wind or other means. Large woody debris forms retention structures in streams and regulates particulate matter transport by stream processes.

Studies in a small (10-ha) western Oregon watershed and adjacent areas have quantified transfer process rates in an example of an old-growth *Pseudotsuga menziesii/Tsuga heterophylla* ecosystem. Excluding debris torrents, total export is estimated to be 500 $kg \cdot ha^{-1} \cdot yr^{-1}$, of which 6 percent is dissolved organic matter, 60 percent dissolved inorganic matter, 8 percent particulate

organic matter, and 26 percent particulate inorganic matter. Total export including debris torrents is 990 $kg \cdot ha^{-1} \cdot yr^{-1}$.

Even excluding from consideration debris torrents, the values for each of these forms of export exceed similar estimates for watershed 6 in the fifty-five-year-old hardwood forest at Hubbard Brook, New Hampshire. Higher export values from the Oregon watershed are mainly a result of its higher annual runoff, more readily weathered bedrock and soil, warmer environments for weathering and decomposition, steeper hillslopes, higher litterfall rates, and greater standing crop of living and dead biomass.

Transfer of inorganic matter appears to be dominated by episodic processes, debris avalanches, and torrents. Solution transfer of inorganic matter is second in importance, followed by suspended sediment and bedload transport in the channel and creep, surface erosion, and root throw from hillslopes.

The relative importance of processes in organic matter transport is less varied. The magnitude of importance of episodic processes suggests that nutrient cycling studies based on short-term records may lead to misleading conclusions concerning long-term ecosystem behavior.

The rate of each process varies in response to ecosystem perturbations as a result of numerous interactions between vegetation and transfer processes. The magnitude and duration of rate increases or decreases vary widely from process to process, depending on the type of ecosystem disturbance. For example, clearcutting eliminates root throw while increasing debris avalanche occurrence by several times over a period of one to two decades. Timing of recovery of various process rates to levels typical of forested conditions is dependent on rates of recovery of key components of vegetation.

Short-term comparisons of transfer rates under clearcut conditions with forested conditions may not yield realistic estimates of management impacts on long-term soil loss. In forests of the Pacific Northwest, long-term erosion history under both natural and human-influenced conditions involves long periods with only minor year-to-year fluctuations interrupted by periods of severe ecosystem disturbance and resulting pulses of accelerated material transfer. Clearcutting and road construction have replaced wildfire as major disturbances of these forests and landscapes. Therefore assessment of management impacts requires knowledge of the frequency and consequences of both management and premanagement disturbances.

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Anadromous Fish Habitat

Fred Everest

<u>OUTLINE</u>

I. Utilization of river networks by anadromous salmonids.

  A. Overview.

  B. The relative importance of big streams verses small streams (recent research from Knowles Creek, Oregon).

    1. The importance of water volume.

    2. The importance of habitat complexity.

    3. Insights for habitat protection and enhancement.

  C. Use of rearing habitats by chinook, coho, and steelhead.

    1. The role of habitat diversity.

    2. The importance of edges.

    3. Insights for habitat protection and enhancement.

II. New insights on quantity and quality of spawning areas.

  A. How much spawning area is enough?

  B. Determination of gravel quality.

  C. The effect of spawning on the quality of gravels.

  D. Insights for protection and enhancement of spawning areas.

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Forest Management and Anadromous Fish Habitat Productivity

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Introduction

In 1976, the USDA Forest Service established a research program to study the biological, physical, and economic aspects of anadromous salmonid habitat. The Anadromous Fish Habitat Research Program is a cooperative effort involving scientists at three USDA Forest Service forest and range research stations: the Pacific Northwest at Portland, Oregon; the Intermountain at Ogden, Utah; and the Pacific Southwest at Berkeley, California.

Scientists at these facilities are studying the relationship between forest management practices and the habitat of anadromous salmonids to develop better ways to achieve concurrent production of timber, fish, minerals, livestock, and other resources. The program is oriented around three types of studies: (1) habitat requirements, (2) effects of various land uses on habitat, and (3) development of ways to improve fish habitat.

The anadromous fishery resources of western North America are produced largely within forested watersheds. Eight species of anadromous salmonids including five salmon—chinook *(Oncorhynchus tshawytscha)*, coho *(O. kisutch)*, sockeye *(O. nerka)*, chum *(O. keta)*, and pink *(O. gorbuscha)*; two trout—steelhead rainbow *(Salmo gairdneri)* and coastal cutthroat *(S. clarki)*; and one char—Dolly Varden *(Salvelinus malma)*, inhabit waters of the Pacific Northwest and Alaska. Habitat requirements of the fish are specific, and alterations of habitat by humans affects production in many ways.

Forest and rangeland management activities that can influence the quality of anadromous fish habitat include timber harvest, road construction, livestock grazing, mining, water developments of various kinds, and recreational pursuits. This paper discusses some of the specific interrelationships between fish habitat and timber harvest, road construction, and livestock grazing. These three activities were selected for discussion because the USDA Forest Service is actively engaged in studies pertaining to these areas of concern.

More specifically the following subjects will be discussed: (1) effects of organic debris and its removal on fish habitat; (2) effects of mass soil movements on fish habitat; and (3) preliminary results of studies concerned with the relationship between different livestock grazing systems and fish habitat.

Organic Debris

Organic debris from forested watersheds of the Pacific Northwest and Alaska enters streams through direct litterfall, various lateral movements including land-

slides, debris torrents, timber felling, and streambank erosion, plus blowdown of trees, treetops, and branches. Natural accumulation of debris in streams is slow and fairly constant in mature forests, and eventually moves toward an equilibrium between the rate of increase and rate of biological and physical processing in old-growth forests (Sedell and Triska 1977). Logging, or change in forest succes due to natural events can shift the equilibrium causing significant chan n streams and fish populations.

Organic debris can be divided into two categories based on size of individual pieces. Large debris consists of tree boles, rootwads, and large limbs, while small debris is composed of needles, leaves, twigs, and branches. The two categories of debris affect the physical characteristics of streams and production of anadromous fish in different ways.

The effects of organic debris on fish production can be either beneficial or detrimental. The effect is often determined by the size of the recipient stream, the size, quantity, and accumulation rate of woody debris entering the channel, and species of fish. Small streams in the Pacific Northwest are strongly influenced by adjacent terrestrial environments and are dependent largely upon external energy inputs, thus tend to be extrinsic and heterotrophic. The following discussion pertains primarily to small streams—first to third order (Strahler 1957)—which are important producers of anadromous salmonids, and yet because of their size are readily influenced by organic debris.

Several major positive effects of organic debris have been identified in previous studies. Large debris creates physical habitat diversity for rearing salmonids (Swanson and Lienkaemper 1978), provides hiding and resting cover in summer (Baker 1979), respites from floods and ice in winter (Bustard and Narver 1975), and stabilizes streambeds and banks (Swanson and Lienkaemper 1978). It also slows downstream movement of inorganic sediments and fine organic matter (Swanson and Lienkaemper 1975), thus providing an energy base for the aquatic food chain and retaining gravels essential for salmonid reproduction and production of fishfood organisms. On the other hand, massive accumulations of large debris can create barriers to migration of anadromous fish (Holman and Evans 1964), cause bank erosion and channel instability during flood events (Helmers 1966), and debris jams when dislodged by high flows can scour streambeds, th removing cover and gravel and altering stream morphology. Small debris pr the primary source of energy for the aquatic food chain in small forested streams (Cummins 1974), but excesses of small debris sometimes cause ponding and depletion of dissolved oxygen in stream waters (Hall and Lantz 1969). Small debris that infiltrates stream gravels can also cause the depletion of intragravel dissolved oxygen (Hall and Lantz 1969) and mortality of incubating salmonid embryos. Accumulations of fine organic material can also produce potentially toxic leachates, particularly in estuaries (Buchanan et al. 1976).

Bryant (1980) measured the evolution of large organic debris after timber harvest in the Maybeso Creek watershed in southeast Alaska. He found a decrease in accumulations of large debris 15 to 20 years after logging, resulting in a decrease in pool areas and an increase in riffles. Although amount of debris decreased in general, remaining debris along the banks and projecting into the channel still influenced channel morphometry, and in some instances contributed to streambank stability. This residual debris helped to maintain pools that were important rearing areas for juvenile salmonids.

Removal of debris has been a concern of resource managers for many years. When should debris be removed and when should it be retained? Often all logging-associated debris has been removed, including large material. In southeast Alaska, natural debris accumulates very slowly (Swanson et al. 1977), and total debris removal often results in a completely open channel (Bryant unpublished). Effects of total debris removal on aquatic invertebrates and Dolly Varden populations were assessed by Elliott (1976). He found a shift in the invertebrate community structure in Starrigaven Creek near Sitka, Alaska, to taxa associated with riffles within a year after debris removal. Dolly Varden populations also exhibited an 80-percent reduction.

Differences between natural and logging debris were also apparent in recent studies in southeast Alaska (Bryant unpublished). Natural debris was often partially decomposed and less concentrated than logging debris. Logging debris was smaller, contained more floatable material, and occurred in more dense and patchy concentrations. Streambed scour and channel instability resulted more from logging debris than from natural debris.

The effects of organic debris may depend on the species of anadromous salmonids present (Narver 1971). Debris that forms pools used by rearing coho salmon may promote sediment deposition in riffles used by pink salmon for spawning. Conversely, removal of debris accumulations may reduce pools and increase riffles, providing less productive rearing areas, but more spawning areas.

Mass Soil Movements

Mass erosion events are a common occurrence in forested steeplands of western North America. Landslides, slump earthflows, and debris torrents account for most mass erosion in the Pacific Northwest but debris torrents are the most universally distributed and frequent events. The rate at which debris torrents occur in steep unstable watersheds is closely linked to timber management and the presence of roads (Swanson and Dyrness 1975). Since anadromous fish and timber are concurrent crops in most western watersheds, it is important to understand how debris torrents affect fish production and harvest.

Debris avalanches, the precursors of debris torrents, are often initiated when intense precipitation saturates exposed soils in road cuts and fills, or clearcuts. The saturated soil slumps under its own weight and forms a slurry which moves rapidly downslope, entraining additional soil, rock, and woody debris. Before losing momentum, most torrents enter the channels of small streams. Once in streams, the soil, rock, and debris might move downstream, scour the channel to bedrock, or be deposited in a massive dam of organic and inorganic debris. In either case, the physical features of the stream channel are changed and fish habitat is altered.

In 1978 we initiated a study to assess the relationships between debris torrents and habitat of anadromous salmonids. The study was designed to: (1) quantify site-specific effects on salmonid habitat immediately below the egress of a debris torrent and assess longevity of those effects, and (2) determine the long-term relationships between debris torrents and fish habitat within an extended reach of stream. The first objective was accomplished with an extensive study of four debris torrents in different watersheds, and the second with an intensive study in a single watershed.

Torrents on four streams in the Oregon Coast Ranges about 50 miles (80 km) west of Eugene were studied from 1978 through 1980. Two of the streams (Hadsall Creek and Knowles Creek) were small at the study sites, flowing less than 0.5 cubic feet per second (cfs) (14 liters/sec.) in summer, and two were larger (Tenmile Creek and Cummins Creek), flowing more than 2.5 cfs (71 liters/sec.) in summer. The smaller streams received large inputs of debris (greater than 1,000 cubic yards (765 m^3)) while the larger streams received smaller amounts (less than 500 cubic yards (380 m^3)). The torrents on Hadsall and Knowles creeks occurred in February 1978, and those on Cummins and Tenmile creeks date from November 1975. The study was designed to assess both spatial and temporal habitat changes caused by debris torrents. Spatial effects were assessed by sampling similar stream habitats above and below a torrent egress (point where torrent exited first order stream and entered second or third order stream) at low summer streamflow and comparing results. Consistent major differences were assumed to be attributable to the effects of debris torrents. Samples collected at given locations over a period of years will also be compared to assess recovery rates.

The major parameters examined included:

1. structure and biomass of fish populations,
2. numbers, biomass, and size of aquatic invertebrates,
3. textural composition of spawning gravels,
4. surface and intragravel dissolved oxygen, and
5. instream cover.

Fish populations examined included cutthroat trout, juvenile steelhead trout, juvenile coho salmon, and freshwater sculpins *(Cottus perplexus* and *C. gulosus).* Numbers and biomass of all species of fish were reduced in all sampling areas impacted by debris torrents (Table 1). Biomass of salmonids was reduced an average of 90 percent in the small streams and 55 percent in the larger streams. Populations of sculpins suffered similar reductions. In three years of sampling, no definite trends toward recovery were noted.

The effects on aquatic invertebrates were variable. Biomass of benthic macroinvertebrates longer than 0.39 inches (1 mm) was lower in square-foot bottom samples collected in torrent-disturbed areas in three of the four streams sampled, but no definite trends in numbers or mean lengths of individual organisms were noted. The largest reduction of benthic invertebrates was observed on the smallest stream, Hadsall Creek.

Texture of spawning gravels and supply of intragravel dissolved oxygen are critical factors in survival of incubating salmonid embryos. Fine sediments less than 0.39-inch (<1-mm) diameter and low concentrations of intragravel dissolved oxygen tend to reduce reproductive success of salmonids. We noted an initial increase in fine sediments of more than 90 percent by weight in torrent-disturbed gravels of Hadsall Creek, and little improvement occurred during two years of sampling (Table 2). Knowles Creek had no upstream gravels near the torrent egress so comparative data for that stream are lacking. No major changes were observed in gravels of the larger streams. Intragravel dissolved oxygen followed a similar pattern, dropping substantially in the disturbed area of Hadsall Creek where sand and organic debris were entrained in gravels, but changing little on the larger streams (Table 2).

Table 1. Biomass of fish in stream habitat above and below a debris torrent egress.

Stream	Species	Biomass, pounds per acre (g per m2)					
		1978		1979		1980	
		Above	Below	Above	Below	Above	Below
Hadsall Creek	Cutthroat	42.97(4.82)	7.93(0.89)	48.85(5.48)	dry	intermittent	dry
	Sculpins	32.36(3.63)	0.36(0.04)	47.24(5.30)	dry	intermittent	dry
Knowles Creek	Coho	4.10(0.46)	2.85(0.32)	4.01(0.45)	0.80(0.09)	dry	5.17(0.58)
	Sculpins	11.86(1.33)	6.86(0.77)	10.70(1.20)	8.65(0.97)	dry	14.00(1.57)
Cummins Creek	Steelhead	9.81(1.10)	3.39(0.38)	31.02(3.48)	6.51(0.73)	21.48(2.41)	4.10(0.46)
	Sculpins	13.28(1.49)	8.74(0.98)	33.42(3.75)	16.58(1.86)	16.40(1.84)	16.31(1.81)
Tenmile Creek	Steelhead	11.32(1.27)	5.61(0.63)	16.76(1.88)	2.94(0.33)	16.49(1.85)	12.75(1.43)
	Sculpins	77.55(8.70)	29.24(3.28)	47.51(5.33)	19.52(2.19)	51.35(5.76)	30.49(3.42)

Table 2. Percent weight fine sediment less than .039-inch (<1 mm) diameter and intragravel dissolved oxygen (mg/l) in salmonid spawning areas above and below a debris torrent egress.

Stream	Parameter	1978		1979		1980	
		Above	Below	Above	Below	Above	Below
Hadsall Creek	Sediment	12.06	23.33	17.16	23.56	—	—
	Oxygen	9.3	6.7	6.5	4.4	6.5	dry
Cummins Creek	Sediment	8.99	6.31	6.73	8.67	—	—
	Oxygen	10.3	8.3	10.7	11.0	8.3	6.1
Tenmile Creek	Sediment	7.26	7.14	8.93	8.54	—	—
	Oxygen	10.0	7.3	10.3	11.7	8.0	6.3

Instream cover was also examined in disturbed and undisturbed areas in the stream channels of Hadsall and Tenmile creeks. The largest changes in cover were observed for the small stream. The torrent scoured most instream cover from the channel of Hadsall Creek below the egress and filled most pools with sediment. Both cover and pool area were reduced by about 70 percent. Pool area on Tenmile Creek was unchanged by the debris torrent, but cover in the form of woody debris increased about 30 percent in the disturbed area.

In summary, the site-specific effect of debris torrents on fish habitat and production was generally negative in the area immediately below the torrent egress. The cumulative effects were greatest on small streams that received large quantities of debris.

Extended Reach Effects—Intensive Evaluation

The effects of debris torrents on habitat and production of anadromous fish were also studied on a 1-mile (1.6-km) reach of Knowles Creek in 1979 and 1980. Observations were also made outside the 1-mile (1.6-km) reference reach. Bedrock in the Knowles Creek watershed is sandstone, and about 80 percent of the stream substrate along the main-stream is composed of sandstone bedrock. In this sediment-poor system, fish habitat diversity in terms of cover, pools, and variations in substrate in the study reach was directly related to organic and inorganic materials deposited in the channel by debris torrents. Habitat utilization by adult and juvenile coho salmon was also strongly associated with habitat changes resulting from debris torrents.

Debris torrents entering first- through third-order stream channels of Knowles Creek appear to have consistent and predictable effects on habitat of anadromous fish. Recent torrents on this stream occurred during intense storm events when all streams in the basin were at or near flood stage. Typically a large mass (>1,000 yd^3(765 m^3)) of large woody debris, rock, and soil entered the stream from a steep side channel. High flows tended to float the woody material out of the system, or deposit it at the stream margins or in a massive debris jam somewhere downstream. Large rock, rubble, and soil in the torrent created a dam in the channel, changed the channel gradient, and created a small lake upstream. Sediment transported by subsequent winter freshets was deposited at the head of the lake, forming new gravel bars. Over a period of years the lakes fill with gravel.

There is evidence of three large debris torrents in the 1-mile study reach of Knowles Creek. Two are recent—one occurred on November 14, 1980, and the other in February 1977. The third is an old torrent of unknown age. Gradient changes caused by the 1977 torrent and the undated torrent were similar. The stream channel rises at 0.8 percent grade in the area of both torrents. A stable dam of boulders and rubble increased the gradient at the egress of the 1977 torrent to 1.6 percent for a distance of 450 feet (137 m), and gradient at the undated torrent was increased to 1.9 percent for 600 feet (183 m). Gradient changes caused by the 1980 torrent have not yet been surveyed. None of the debris dams retarded upstream passage of anadromous salmonids. The 1977 and 1980 torrents created lakes with volumes of 1.7 and 1.8 acre feet (2,097 and 2,220 m^3), respectively. The capacity of the debris torrent ponds for rearing underyearling coho salmon exceeded other habitats available in the stream. The population of coho rearing

during the summer in the pond created by the 1977 torrent was estimated at 875 fish in 1979 and 965 in 1980, or an average of about 1.77 fish per lineal foot (5.8 fish per lineal meter) of stream channel. Coho population estimates from four unponded segments of Knowles Creek indicated an average of 0.18 coho per lineal foot (0.6 per m) in summer. In 1979 and 1980, the debris torrent pond was rearing under-yearling coho at a rate nearly 10 times greater than the unponded channel. The number of coho rearing in the 530-foot-long (160-m) pond was estimated to be approximately equal to the number rearing in 1 mile (1600 m) of unponded habitat.

The debris torrent ponds also provided habitat for adult coho. Ponds created by the torrents provided resting habitat for adult coho awaiting spawning. Ponds also provided natural settling basins for gravels transported downstream during freshets. Observations made during the winter of 1980 illustrate the importance of torrent ponds to adult coho. On December 13, 1980, 92 adult coho salmon were observed in the 1-mile (1.6-km) study reach of Knowles Creek. Eighty-five percent of the fish were either resting in the deep ponds created by the 1977 and 1980 torrents, or spawning on gravels impounded by the three torrents. Ten coho redds were counted in the reach and 7 were in gravels impounded by torrent-produced dams.

Coho make immediate use of habitat created by debris torrents. The 1980 torrent was less than three weeks old when 14 adults were seen resting in the new pond and 2 redds were observed on 25 yd^2 (21 m^2) of freshly impounded gravel at the head of the pond.

Our study of an extended reach of stream on Knowles Creek has helped clarify the effects of debris torrents on fish habitat in that watershed. Although torrents have a negative effect on habitat in areas inundated with organic and inorganic debris, the over-all effect on Knowles Creek appears positive. Within the extended reach, debris torrents enhanced spawning and rearing habitat for adult and juvenile coho salmon and did not interfere with adult migration routes.

The results obtained on Knowles Creek might be generally characteristic of other sediment-poor systems in the Coast Range sandstone formation. Similar changes in habitat probably do not occur in sediment-rich systems where sediment added by torrents would add little to habitat diversity and torrent-created ponds would fill rapidly with sediment. Additional research is needed to determine how broadly results from Knowles Creek can be inferred.

Future research is also needed to determine how the accelerated occurrence of debris torrents caused by timber management affects fish habitat. The rate of accumulation of debris in stream channels could be the determining factor in whether cumulative effects on habitat of anadromous salmonids are positive or negative.

Livestock Grazing Systems

Livestock use streamside areas heavily, for feeding and resting. The stream itself is often their only source of water; this also promotes use of streamside areas. Many studies have described the adverse impacts of overgrazing and concentration of livestock along streams, but these have usually dealt with physical impacts such as soil compaction, streambank trampling, and reduction of streamside vegetation (Meehan and Platts 1978). A few studies have compared biomass

of salmonids in stream reaches protected from streamside grazing with reaches along which grazing occurred. Gunderson (1968) showed that brown trout (*Salmo trutta* Linnaeus) production was considerably greater in sections of Rock Creek, Montana, adjacent to ungrazed areas than in sections adjacent to grazed areas. These findings were later verified by Marcuson (1971). Platts and Rountree (19[illegible]) reported fish habitat was damaged, primarily from bank trampling, more along sections of Bear Valley Creek, Idaho, than along ungrazed sections.

Several different livestock grazing systems are in use today (Meehan and Platts 1978). These systems differ primarily in the intensity of use within various pastures of a grazing allotment. All pastures may be grazed continuously throughout the grazing season; a given pasture may be grazed during the early part of the season and left ungrazed during the latter part, or vice versa; various pastures may be rested periodically for a full grazing season; or any number of combinations or modifications of these strategies may be used.

The USDA Forest Service Anadromous Fish Habitat Program has two studies underway to evaluate the effects of various grazing systems on fish habitat. One of the studies is being conducted on several streams in central Idaho; the other is on Meadow Creek in northeastern Oregon's Blue Mountains. Neither study has been completed, so conclusive results cannot be reported at this time. We have been working on the Meadow Creek study, and will briefly discuss this study and some preliminary observations.

The stream was divided into four 1.25-mile (2.0-km) sections to study effects of various livestock grazing systems, with and without wildlife (deer and elk) influence, with a separate 1.25-mile upstream "control" section. The fisheries portion of the study concentrated on the upstream treatment section. This section was subdivided into five units. The first year, cattle grazed season-long in the unit farthest downstream, while upstream units remained ungrazed. The second year, the two downstream units were grazed, etc., so that at the end of the five-year study, the most downstream unit had been grazed for five years, the next upstream unit for four years, etc. with the farthest upstream unit being grazed for one year. Stream channel and bank profiles, benthic and invertebrate drift samples, and steelhead trout population estimates and stomach contents were obtained before cattle were put on, at mid-season, and after cattle were removed each year. W[illegible] temperature and various chemical parameters were measured as well.

Data analysis will continue for about two more years. Presently, nothing can be said about effects of grazing on invertebrate organisms. Preliminary observations show no obvious differences between fish standing crops in the control section and the treatment sections after three years of season-long grazing. Stream channel changes have not yet been fully analyzed, but preliminary examinations indicate changes are due more to flow conditions than to direct livestock effects.

The primary purpose of this discussion has been to outline the general scope of the grazing study and the kinds of data that will result.

Summary

The USDA Forest Service Anadromous Fish Habitat Research Program is investigating relationships between forest and rangeland management and fish habitat productivity. Scientists are studying habitat requirements of anadromous salmo-

nids, the effects of various land uses on habitat, and habitat enhancement techniques. Current research is clarifying the impact of large woody debris, mass soil movements, and livestock grazing systems on habitat productivity for anadromous salmonids.

Large woody debris can create habitat for rearing salmonids, but may cause sedimentation in spawning areas. Large, naturally occurring debris can promote streambank stability and reduce streambed scour. Large accumulations of fine organic debris can adversely affect salmonid habitat by reducing dissolved oxygen and producing toxic leachates. In some instances, large debris accumulations may impede fish movement. Total removal of debris can result in a completely open channel, promoting streambed scour, streambank instability, and loss of fish habitat productivity.

Debris torrents, a common mass erosion event in the Pacific Northwest, have a negative impact on habitat and production of anadromous salmonids in small streams immediately downstream from the torrent egress. In a restricted area, both spawning and rearing habitat are degraded and fish production is reduced. Studies within a 1-mile (1.6-km) reach of Knowles Creek, however, indicate that the total effect of debris torrents in that sediment-poor watershed tends to be positive. Torrents created habitat diversity by adding boulders, rubble, gravel, and woody debris to the channel and increased both quantity and quality of habitat for juvenile and adult coho salmon. Torrents are a natural physical process that provide woody debris and gravel necessary to maintain productive fish habitat. The rate at which debris torrents occur might be the most important factor in determining whether cumulative effects are positive or negative.

Very preliminary results of our livestock grazing study do not show profound effects on fish populations among various grazing systems or between one to three years of season-long grazing and ungrazed controls. Final analysis of results should be completed in 1982.

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Distribution and Abundance of Rearing Salmonids in Relation to Water Volume in an Oregon Watershed

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Abstract

Knowles Creek, a 5th order tributary of the Siuslaw River, drains a 58 km^2 basin in the Tyee Sandstone formation of west central Oregon. The stream heads at an elevation of 950 m and flows into the Siuslaw estuary. The basin has an extensive history of timber harvest dating back to the late 1800's. Most old-growth timber has been removed from riparian zones and the main channel was used historically for water transportation of logs. Consequently, large woody debris and other features contributing to channel roughness and complexity are sparsely distributed in the system.

Knowles Creek contains seven species of fish. The most common are coho salmon (*Oncorhynchus kisutch*) and freshwater sculpins (*Cottus* sp.), followed in abundance by cutthroat trout (*Salmo clarki*), steelhead trout (*S. gairdneri*), blackside dace (*Rhinichthys osculus nubilis*), and redside shiner (*Richardsonius balteatus*). Salmonids and cottids are distributed throughout the basin while cyprinids are confined to the middle and lower basin.

In 1981 we studied the relationship between water volume and standing crop of fish in the Knowles Basin. We divided the watershed into upper, middle, and lower sections, each with an area of about 19 km^2 and containing between 8.4 and 9.0 km of fish-producing stream. The distribution and abundance of rearing salmonids was determined in each section during the period of minimum streamflow. Total water volume in each section was also quantified at low flow and both biomass per unit area and total populations of fish were estimated in each section.

The frequency of pools was highest in headwater tributaries of the upper section (133 pools/km) and generally decreased downstream (45 pools/km in the lower section). The average volume of pools increased from 1.3 m^3 in tributaries of the upper section to 46.1 m^3 in the lower section. Between 92 and 98 percent of water volume at low flow was in pools. Five percent of the total volume was in the upper section, 28 percent in the middle, and 67 percent in the lower section. Mass erosion events in the basin and beaver dams contributed significantly to stored water volume.

Biomass of juvenile coho salmon was highest in small tributaries of the upper section (30 g/m^3) and positively correlated with pool volume. Coho biomass decreased steadily downstream to about 8 g/m^3 in the lower section and was more strongly influenced by pool complexity than pool volume. Cutthroat trout biomass was highest in the middle section and dace and shiner biomass was highest in the lower section. Condition factors of fish were not significantly different between sections, but fish of equal age were largest in downstream waters.

While the highest biomass of fish per m^3 of water was observed in small headwater tributaries, large pools in the lower section contained the greatest absolute numbers of fish. Forty-five percent of the coho salmon rearing in the basin were in the lower section, 35 percent in the middle, 12 percent in the upper, and 8 percent in the tributaries.

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Our results indicate that fish habitat management plans derived from reach-specific sampling schemes can underestimate the importance of lower mainstem streams for salmonid production. Also, efforts to enhance rearing habitat are probably most effective in larger downstream waters where greater absolute numbers of salmonids can be produced.

Current enhancement efforts for anadromous salmonids in much of the western United States emphasize removal of migration barriers and habitat manipulation on small headwater streams. Our research plus investigations of the historic character of streams draining forested watersheds indicates that salmonid production in pristine systems was concentrated in larger streams (> 3rd order) and that the present fixation with enhancement on headwater streams will not restore salmonid populations to historic levels.

James R. Sedell

Acquisition and Utilization of Aquatic Habitat Inventory Information

Proceedings of a Symposium held 28–30 October 1981, Portland, Oregon

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NEW PERSPECTIVES ON SAMPLING, ANALYSIS, AND INTERPRETATION OF SPAWNING GRAVEL QUALITY[1]

F. H. Everest, F. B. Lotspeich, W. R. Meehan[2]

Abstract.--Methods of sampling salmonid spawning gravels and interpreting their quality have improved in recent years. Multiple probe freeze-core samplers allow vertical subsampling of gravel cores and provide an improved understanding of the textural composition of gravels. Vertical subsampling of cores has shown that textural composition of gravels varies with depth below the substrate surface and is changed by the hydraulic forces exerted by spawning fish. These findings have important management implications.

INTRODUCTION

The amount and condition of streambed gravels available to spawning salmonids has been a perennial concern of fishery managers. Biologists conducting habitat inventories routinely record quantities of gravel available to spawning salmonids and estimate the quality of gravels based on surface appearance. Resource managers often sample streambed gravels to monitor changes in textural composition and quality resulting from land management activities or changes in streamflow regimen caused by water development projects. Monitoring is also conducted to assess changes in substrate composition resulting from watershed rehabilitation projects. Finally, gravel sampling is frequently used to determine the success of salmonid habitat enhancement projects designed to increase the quantity and quality of spawning areas.

Gravels containing a low proportion of fine sediments have long been recognized (Harrison 1923) as a critical requirement for successful salmonid reproduction. Significant advances in equipment and techniques for quantitative sampling of gravels, however, have been made only within the past few years. New methods for assessing gravel quality have also been developed recently. The new equipment, procedures, and techniques have shown that the textural composition of streambed gravels changes with depth below the substrate surface (Everest et al. 1980, Scrivener and Brownlee 1981). The new insights on vertical textural variations of spawning gravels have several important management implications. Our objective is to describe the recent advances in gravel sampling and analysis techniques and their implications for management, and to recommend current state-of-the-art procedures for assessing gravel quality.

GRAVEL SAMPLING

Sampling Equipment

Two basic types of gravel samplers are presently in use, the "McNeil" type sampler and the freeze-core sampler. Both devices have advantages and disadvantages that are described below.

McNeil Sampler

The first attempts at quantitative samplers for general use consisted of metal tubes, open at both ends, that were manually forced into the substrate. Sedimentary material encased by the tubes was removed by hand for analysis. A variety of samplers using this principle have been developed, but one described by McNeil and Ahnell (1960) has become widely accepted for sampling streambed sediments.

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[1]Paper presented at the Symposium on Aquisition and Utilization of Aquatic Habitat Inventory Information. [Portland, Oregon, October 1981].

[2]Fred H. Everest is Research Fishery Biologist with the USDA Forest Service, Pacific Northwest Forest and Range Experiment Station, Corvallis, Oregon. Frederick B. Lotspeich is presently a volunteer to the USDA Forest Service. Formerly he was Research Physical Scientist, U.S. Environmental Protection Agency, Corvallis, Oregon. William R. Meehan is Project Leader, Anadromous Fish Habitat Research, USDA Forest Service, PNW, Juneau, Alaska.

The McNeil sampler is usually constructed of stainless steel and can be modified to fit most sampling situations. It is lightweight and portable and can be used in remote locations if necessary. Disadvantages of the sampler are:

(1) Core materials are completely mixed upon removal so no interpretation can be made of vertical or horizontal differences in particle size distribution within a core, (2) sampling depth is limited by such factors as water depth and length of the collector's arm, (3) the core tube often pushes larger particle sizes out of the collecting area, (4) suspended sediments in the core are lost, (5) sediment particle sizes larger than the core tube cannot be collected, and (6) often the sampler can not be inserted to the required depth if sediment particles are large or if the substrate is compacted.

Regardless of the many limitations of this sampler, when time and costs are considered, it is probably the most economical method available for establishing rough estimates of substrate particle size distribution.

Freeze-Core Samplers

In the last decade scientists have been experimenting with cryogenic devices to obtain sediment samples. These devices, generally referred to as "freeze-core" samplers, consist of a hollow probe driven into the streambed and cooled with a cryogenic medium. After a prescribed time of cooling, the probe and a frozen core of sediment adhering to it are extracted. Liquid nitrogen, liquid oxygen, solidified carbon dioxide ("dry ice") and acetone, dry ice and alcohol, and liquid carbon dioxide (CO_2) have been used experimentally as freezing media. Several years of development produced a reliable single-probe sampler (Walkotten 1976) that uses liquid CO_2.

The accuracy and precision of the single probe freeze-core and McNeil sampler have been compared in laboratory experiments.[3] Samples collected by both devices were found to be representative of a known sediment mixture, but the freeze-core sampler was more accurate (Walkotten 1976). An important advantage of cryogenic samplers is that frozen cores can be vertically subsampled. Freeze-core samplers are also more versatile, functioning under a wider variety of weather and water conditions.

Cryogenic samplers also suffer several disadvantages. It is difficult to drive probes into substrate containing many particles over 10 cm in diameter, and the freeze-core technique is equipment intensive, requiring CO_2 bottles, hoses, manifolds, probes and a sample extractor. Also, since it is necessary to vertically subsample cores by depth for accurate interpretation of gravel quality (Everest et al. 1980), it is often necessary to collect more massive cores than can be easily obtained by the single-probe technique. For example, Adams (1980) used a single-probe device to extensively sample stream substrates in the Oregon Coast Ranges. He was able to extract up to six cores of sediment averaging about 1.6 kg/core per 9-kg (20-pound) tank of CO_2. Individual cores of such size are minimal for vertical subsampling. Skaugset (1980), on the other hand, collected cores exceeding 20 kg with a single-probe device using 10 liters of liquid nitrogen per sample. Skaugset's samples were large enough for representative subsampling, but liquid nitrogen is more expensive, difficult to obtain, store, and use than liquid CO_2.

To alleviate problems caused by the small size of sediment cores obtained by single-probe samplers using CO_2, and to avoid use of liquid nitrogen as a cooling medium, Lotspeich and Reid (1980) and Everest et al. (1980) have modified the single-probe device. The modified freeze-core sampler uses a triangular array of three probes driven into the substrate through a template that keeps the probes in fixed relationship to each other. The "tri-tube" sampler (fig. 1) retains all the advantages of the single-probe freeze-core sampler, but extracts larger cores—often more than 20 kg per 9-kg (20-pound) tank of CO_2--which are more representative of substrate composition than small cores obtained by the single-probe sampler, or cores obtained with McNeil samplers.

We recommend use of the multiprobe procedure if an analysis of horizontal and vertical stratification of sediments is required. We suggest use of the tri-tube sampler described by Lotspeich and Reid (1980) and Everest et al. (1980) when numerous cores must be collected, and the Platts-Penton (1980) sampler when only a few large cores are needed.

The freeze methods allow collection of eggs and alevins in a redd at any stage of development, will function at most air or water temperatures or stream depths, and allow analysis of horizontal and vertical location of the eggs and alevins. Because these techniques require several pieces of equipment, they are most conveniently used in accessible areas.

SAMPLING LOCATION AND DEPTH

Selection of spawning sites by salmonids is a nonrandom activity. Adult salmonids seeking locations to spawn respond to environmental variables such as water depth and velocity, substrate composition, and proximity to cover. Because both sediment particle-size distribution and redd site selection are nonrandom events, the location from which samples are drawn to

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[3]Koski, K. Victor, and William J. Walkotten, unpublished data on file at the Forestry Sciences Laboratory, Corvallis, Oregon.

characterize spawning gravels should be identified by an experienced fishery biologist. Samples should only be drawn from locations that meet the known spawning requirements of a species. The suitability of each sampling site should be determined by quantitative measurements of water depth and velocity. The depth at which the sample is extracted is also critical to the analysis. Samples should be taken only as deep as the average depth of egg deposition for the species being studied. Since there is substantial stratification in stream gravels, sampling above or below the level of egg deposition might yield an inaccurate estimate of the size and distribution of particles within a redd. If prediction of survival to emergence of salmonid fry is desired, all samples should be collected from redds just prior to onset of emergence. Otherwise, temporal variations in gravel composition (Adams and Beschta 1980) might lead to inaccurate assessments of gravel quality at the onset of emergence.

GRAVEL ANALYSIS

Analysis of substrate samples is accomplished by sorting sediments through a series of sieves, determining the fraction of each pre-specified size group of sediment in the sample, and making an inference about the quality, or changes in quality, of the substrate for salmonid reproduction.

Sorting Gravels

Two primary methods for sorting sediments are the "wet" method and the "dry" method. The wet method can be done in the field but is the least accurate because water is retained in pore spaces of the sediments. The wet method uses a water flushing technique with some hand shaking to sort sediment through sieves. The trapped sediment on each screen is allowed to drain and is poured into a graduated water-filled container. The amount of water displaced determines the volume of that size fraction plus the volume of water retained in the pore spaces of the sediment. A correction factor (see Shirazi et al. 1981) must be applied to each size fraction of sediment to account for retained water.

For more exacting results, samples should be analyzed by the dry method. Samples are transported to the laboratory, oven or air dried, and sorted through sieves with a mechanical shaker. The proportion of individual size fractions in a sample is then determined gravimetrically. We recommend the Wentworth sieve series which is a geometric progression of 12 size-classes ranging from 0.062 to 100 mm (0.002 to 3.94 in). The upper limit might seem arbitrary, but it approximates the largest size particles in which most salmonids will spawn. Consequently, few grains larger than 100 mm are present in preferred spawning areas.

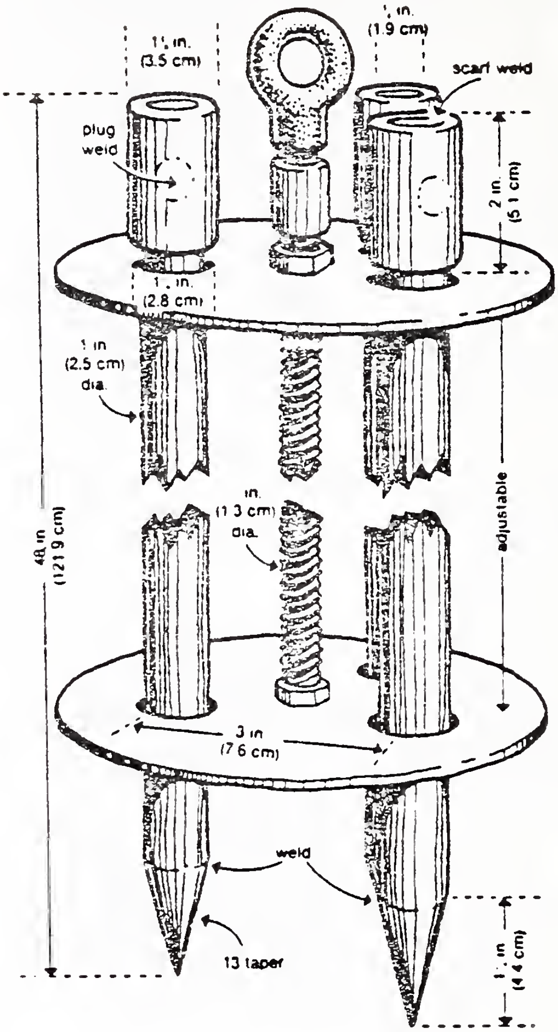

Figure 1.—Schematic diagram of the tri-tube cryogenic gravel sampler.

SELECTION AND USE OF QUALITY INDICES

The quality of gravels for salmonid reproduction has traditionally been estimated by determining the percentage of fine grains (less than some specified diameter) in samples collected from spawning areas. Field data have often been compared to results of laboratory studies (e.g. Phillips et al. 1975) to estimate survival to emergence of various species of salmonids. While an inverse relationship between percent fines and survival of salmonid fry has been demonstrated by several researchers, beginning with Harrison (1923), use of percent fines alone to estimate gravel quality has a

major disadvantage. Use of only fine grains of sediments to characterize the quality of a sample ignores the textural composition of the remaining particles which can have a mitigating effect on survival. For example, imagine two samples each containing 20-percent by weight fine sediment particles less than 1-mm diameter, but the average diameter of larger particles is 10 mm in one sample and 25 mm in the other. Interstitial voids in the smaller diameter material would be more completely filled by a given quantity of fine sediment than voids in the larger material and the subsequent effect on survival of salmonid fry would be very different. Percent fines is a reasonable index of gravel quality but has serious limitations because it ignores the textural composition of the remainder of the sample.

Other quality indices have recently been developed in an attempt to improve upon the percent fines method. Platts et al. (1979) used the geometric mean diameter (d_g) of sediment particles in a sample as an index of salmonid incubation success. This has advantages over the commonly used percent fines method in that (1) it is a conventional statistical measure used by several disciplines to represent sediment composition, (2) d_g relates to the permeabililty and porosity of channel sediments and to embryo survival as well as or better than percent fines, and (3) d_g is estimated from the total sediment composition. Despite these advantages, d_g has been shown by Beschta (1981) to be rather insensitive to changes in stream substrate composition caused by sediment from roads in a Washington watershed. Also, Lotspeich and Everest (1981) have shown that use of d_g alone can lead to erroneous conclusions concerning gravel quality. Because of these problems, Beschta (1981) has raised serious questions regarding the utility of geometric mean diameter as a quality index when used as the sole criterion.

Tappel (1981) offers another approach which is a modification of the d_g method and uses a linear curve to depict particle size distribution by assigning the points 0.8 mm (0.03 inches) and 9.5 mm (0.37 inches) for determining a line. According to Tappel the slope of the line gives a truer picture of fine sediment classes detrimental to incubation. A major drawback of this procedure, as with percent fines, is that it ignores the larger particles in a sample and might, consequently, suffer the same limitations.

A quality index which appears to overcome limitations of percent fines and geometric mean has been reported by Lotspeich and Everest (1981). Their procedure uses a measure of the central tendency of the distribution of sediment particle sizes in a sample and the dispersion of particles in relation to the central value to characterize the suitability of gravels for incubation and emergence of salmonids. These two parameters are combined to derive a quality index called the "fredle index," which provides an indicator of sediment permeability and of pore size. The measure of central tendency used is the geometric mean (d_g). In addition to d_g, the size distribution of sediment particles in a sample is a useful descriptor of a gravel's reproductive potential for salmonids. To quantify the distribution of grain sizes in gravels, Lotspeich and Everest (1981) have used the sorting coefficient (S_o) described by Krumbein and Pettijohn (1938). S_o is derived by taking the square root of the quotient of the grain size at the 75th percentile divided by that at the 25th percentile. Permeability and pore size, which control movement of water and alevins through gravel, are determined largely by the size distribution of grains in a sample. These two substrate parameters are the primary legislators of survival-to-emergence of salmonid embryos.

The Fredle index (f) is calculated by the following method:

$$f = \frac{d_g}{S_o};$$

where, $d_g = (d_1^{w_1} \times d_2^{w_2} \times \ldots\ldots\ldots d_n^{w_n})$,

d_n = mid-point diameter of particles retained on the nth sieve,

w_n = decimal fraction by weight of particles retained on the nth sieve, and

$S_o = \sqrt{\frac{d_{75}}{d_{25}}}$ = sorting coefficient--

d_{75} and d_{25} = particle size diameters at which 75 and 25 percent, respectively, of the sample is finer on a weight basis.

Fredle numbers for sediment with a single grain size will be equal to the geometric mean because S_o is then 1. Sediments with the same d_g will have f numbers less than the geometric mean as S_o increases. Sediments with small d_g values are less permeable than those with larger means because pores are small and intragravel flow and movement of alevins is impeded even though S_o might be 1. Also sediments with large d_g might be slowly permeable when S_o is large because pore spaces are occupied with smaller grains that impede interstitial flow and movement. Thus, the magnitude of the Fredle index numbers is a measure of both pore size and relative permeability, both of which increase as the index number becomes larger.

The Need to Subsample Substrate Cores by Depth

Use of freeze-core samplers has clearly demonstrated that particle size distribution of gravels can vary with depth below the substrate surface (Everest et al. 1980, Scrivener and Brownlee 1981). Our research has shown that the

quality of gravels for incubation of salmonid eggs and emergence of alevins generally decreases with depth below the surface of the substrate. The difference between quality of surface layers and subsurface layers appears to be directly related to the load of fine sediment transported by the stream.

In 1978 and 1979 we collected substrate samples from salmonid spawning areas on four streams in the Rogue River basin of southwest Oregon. Two streams support large populations of fall chinook salmon (*Oncorhynchus tshawytscha* (Walbaum)) and two support large runs of summer steelhead (*Salmo gairdneri* Richardson). Each pair of streams was selected because one member of the pair carried a much higher load of fine sediment during freshets than the other. Samples collected from the streams utilized by chinook were analyzed by 10-cm increments from the surface to a depth of 30 cm. Samples from spawning areas of steelhead were analyzed by 7.5-cm increments to a depth of 30 cm. The results of this analysis (tables 1 and 2) demonstrate that the quality of gravel for salmonid reproduction often decreases substantially with depth below the substrate surface. Such knowledge indicates that accurate inferences of gravel quality can only be made from samples that have been partitioned and analyzed by depth increments.

How to Subsample Substrate Cores by Depth

A major advantage of the freeze-core sampler is that it provides opportunity for vertical subsampling of substrate cores. Everest et al. (1980) have developed a subsampler consisting of a series of open-topped boxes made of 26-gage galvanized sheet metal (fig. 2). A core is laid horizontally on the boxes of the subsampler and thawed with a blow-torch. Sediments freed from the core drop directly into the boxes below. Individual subsamples can then be dried, sorted, and analyzed for textural composition and quality.

EFFECTS OF NATURAL GRAVEL STRATIFICATION ON INTERPRETATION OF GRAVEL QUALITY

Our investigation of the characteristics of spawning gravels using the tri-tube freeze-core sampler and vertical core subsampler have yielded some important implications for future analyses of stream substrates and interpretation of past work. Three major implications are apparent:

1. Surface appearance of gravels is an inadequate and often misleading indicator of gravel quality for salmonid reproduction although stream surveyors often estimate the quantity and quality of available spawning gravels by visual

Table 1.--Changes in textural composition and quality of gravels in chinook salmon spawning areas as related to depth below the substrate surface, Evans Creek and Slate Creek, Rogue River basin, Oregon, 1979.

Stream	Sample depth (cm)	Geometric mean (mm)	Percent fines <1 mm	Fredle index
Evans Creek,	0-10	11.2	12.1	3.6
high sediment	10-20	7.6	22.0	1.5
stream (n=5)	20-30	2.5	42.5	0.4
Slate Creek,	0-10	13.8	6.5	5.7
low sediment	10-20	13.0	7.5	5.1
stream (n=5)	20-30	12.7	12.5	4.4

Table 2.--Changes in textural composition and quality of gravels in steelhead spawning areas as related to depth below the substrate surface in Foots Creek and Sams Creek, Rogue River basin, 1979.

Stream	Sample depth (cm)	Geometric mean (mm)	Percent fines <1 mm	Fredle index
Sams Creek,	0-7.5	9.4	12.3	3.8
high sediment	7.5-15	7.3	16.4	2.7
stream (n=6)	15-22.5	9.1	14.0	2.9
	22.5-30	9.6	13.1	3.2
Foots Creek,	0-7.5	14.5	6.9	9.4
low sediment	7.5-15	10.8	10.8	3.1
stream (n=6)	15-22.5	11.6	12.1	3.1
	22.5-30	13.3	11.2	5.2

appearance of the substrate surface. In addition to the visual assessment, a boot heel is often ground into the substrate to test the compaction or concretion of gravels as an index to quality. Neither method, however, adequately describes gravel quality because gravel quality cannot be assessed at the 20-to 30-cm depth where spawning salmonids usually deposit eggs.

Our research has shown that streambed gravels with similar surface properties and appearance can have very different properties a few centimeters below the surface. For example, surface appearance of gravels in Foots Creek and Evans Creek in the Rogue River basin, Oregon, are very similar and would be classified "good" by visual inspection. Geometric mean particle diameter, sorting coefficient, Fredle quality index, and percent fines <1-mm diameter are also very similar within the top 10-cm layer of gravel in each stream (table 3). The physical characteristics of gravels, however, diverge markedly in the 10-to 20 and 20-to 30-cm strata (table 3). If salmonid fry were forced to emerge through the 20-to 30-cm strata on Evans Creek which contained 41.5 percent fines <1-mm diameter and a Fredle index of 0.4, low survival would be expected. Foots Creek, on the other hand, contained only 11.5 percent fines <1-mm diameter and had a Fredle index of 3.6 in the 20-to 30-cm depth strata. More than 50-percent survival to emergence would be expected under the latter conditions.

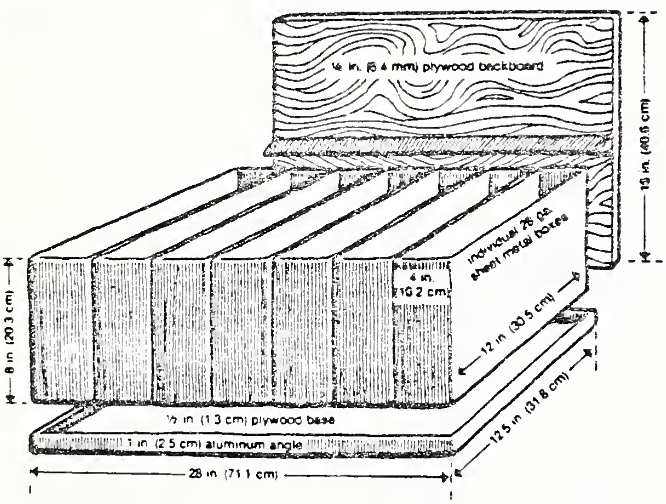

Figure 2.--Diagram of freeze-core subsampler.

2. Failure to stratify cores vertically into subsamples can also misrepresent quality of gravels for salmonid reproduction. Gravel sampling equipment that mixes the contents of a gravel core during extraction, for example, the McNeil sampler, can result in quality estimates either higher or lower than conditions actually faced by emerging fry. For example, if chinook salmon on Evans Creek deposit eggs 30 cm below the substrate surface and 30-cm-deep cores, mixed during extraction, are removed from redds and analyzed, the predicted quality of gravels in the redd exceeds actual conditions faced by fry during emergence (table 4). Data from mixed cores yield an average estimate of 7.6 percent fines <1-mm diameter and a Fredle index of 6.0 in chinook redds. Data from samples taken by freeze-core and stratified into 10-cm increments indicate a Fredle quality index of 3.3-and 10.8-percent fines <1-mm diameter are located between 20 and 30 cm below the substrate surface. The latter are the actual conditions fry must traverse during emergence, not the conditions calculated after mixing the contents of the core.

The quality of gravels in redds can also be underestimated by failing to vertically subsample cores. If core samples collected from redds include a layer of gravel and fine sediment below the level of salmonid eggs and the contents of the core are mixed during extraction, the result is usually an underestimation of gravel quality when compared to samples that include only gravels above egg level. Our research has shown that spawning can remove 20 percent or more of the fine sediments <1-mm diameter in redds. Eggs are deposited at the lowest level of the "cleaned" gravels. If, for example, chinook salmon deposit eggs about 30 cm below the substrate surface, then cores collected with McNeil samplers should not exceed the 30-cm depth. Collection of 40-cm deep gravel cores that include 10 cm of uncleaned gravel below the eggs result in depressed estimates of gravel quality within a redd. Samples collected from Evans Creek in the Rogue River basin illustrate this point. Forty-cm-deep samples were collected with freeze-core equipment and analyzed by 10-cm-depth strata. Strata in individual cores were then combined to compare gravel quality in 30-and 40-cm columns of the same cores (table 5). When samples were mixed and 10-cm of gravel below egg level was included, there was an apparent decrease in gravel quality within the redds. Neither 30-nor 40-cm mixed cores, however, provide an accurate estimate of conditions that fry must actually penetrate during emergence. The actual gravel quality between the 20-and 30-cm depth (table 5) was substantially lower than estimates from combined strata in a 30-cm core, but was in this example coincidentally similar to combined strata for a 40-cm core.

Table 3.--Comparison of gravel composition and quality in surface and subsurface layers in spawning areas of Evans Creek and Foots Creek, Rogue River basin, Oregon.

	Evans Creek Sample depth (cm)			Foots Creek Sample depth (cm)		
Parameter	0-10[1]	10-20	20-30	0-10[1]	10-20	20-30
Geometric mean (mm)	15.7	5.6	2.6	16.9	9.3	11.8
Percent fines <1 mm	6.2	23.1	41.5	5.4	10.6	11.5
Fredle index	8.0	1.3	0.4	8.4	3.0	3.6

[1] Characteristics of surface layers are very similar and look alike.

Table 4.—Comparison of gravel texture and quality in samples (n=6) taken from chinook redds and analyzed first by depth strata and then mixed and analyzed as a unit, Evans Creek, Rogue River basin, Oregon.

		Sample depth (cm)		
Parameter	0-10	10-20	20-30[1]	0-30 mixed
Geometric mean (mm)	25.1	10.7	9.0	13.3
Percent fines <1 mm	1.9	8.7	10.8	7.6
Fredle index	16.3	4.3	3.3	6.0

[1] Fry must actually traverse these conditions to emerge rather than conditions indicated by the 0-to 30-cm mixed core.

Table 5.--Comparison of gravel texture and quality in samples (n=6) including materials from above egg level (0-30 cm) in chinook redds, and below (30-40 cm), Evans Creek, Rogue River basin, Oregon.

	Sample depth (cm)					
	Above eggs			Below eggs		
Parameter	0-10	10-20	20-30	30-40	0-30 mixed	0-40 mixed
Geometric mean (mm)	25.1	10.7	9.0	5.0	13.3	9.3
Percent fines <1 mm	1.9	8.7	10.8	31.1	7.6	13.4
Fredle index	16.3	4.3	3.3	1.7	6.0	3.3

3. Laboratory studies of survival to emergence of salmonid fry utilizing artificial gravel mixtures are not very useful for predicting survival to emergence in the field. Gravel mixtures produced in the laboratory often contain only a few graded sediment particle sizes both for convenience and to allow standardization of mixes. Gravels with such simple textural composition are usually not found in the field. Secondly, eyed eggs are usually planted at a specified depth (e.g. 25 cm) in the homogeneous lab mixtures. Since our studies indicate that the texture of stream gravels usually changes with depth, it is difficult to make direct comparisons of emergence success between lab and field studies. In a study of the effects of sand on emergence of coho salmon (O. kisutch (Walbaum)) and steelhead trout, Phillips et al. (1975) used an artificial control gravel mixture of six particle size groups ranging from 32 to 3 mm with some intermediate size groups missing. Varying amounts of 1-to 3-mm-diameter sand were added to the control mixture; no particles less than 1-mm diameter were added. Eyed eggs were planted at a depth of 25-cm in each homogeneous mixture and survival to emergence was monitored. Survival was inversely related to the amount of sand in the mixtures.

Examining the results from just one of the mixes (20 percent sand) used by Phillips et al. (1975) will illustrate the problems associated with inferring lab data to the field. One group

of alevins was forced to emerge through a 25-cm-thick mixture of homogeneous gravel containing 20 percent sand. No such homogeneous columns 25-cm-deep have been observed in our field studies, although samples mixed during removal often contain an average of 20 percent 1-to 3-mm sand. Field samples containing an average of 20 percent 1-to 3-mm sand when subsampled, however, revealed that textural composition was changing rapidly with depth. A field sample containing 20 percent sand mixed during extraction might seem similar in character to the 20-percent-sand lab mix unless the sample is subdivided by depth for analysis (fig. 3). Subsampling, however, might reveal that fry actually must traverse a layer of gravel containing more than 40 percent sand at the 20-to 30-cm depth, while the 0-to 10-cm depth might contain less than 10 percent sand.

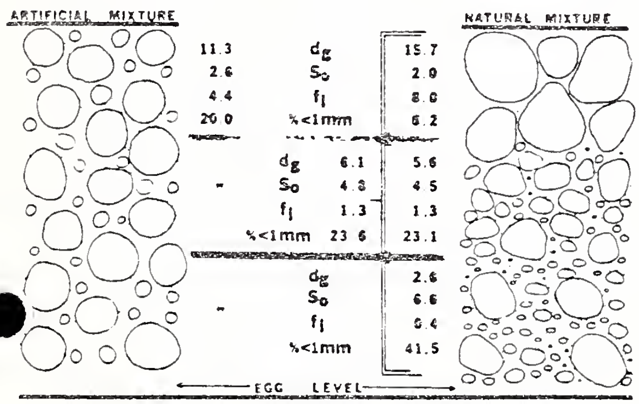

Figure 3.--Diagramatic comparison of the characteristics of artificial and natural gravel mixtures. Because most lab mixtures fail to simulate natural gravels, caution must be used when applying results of lab survival studies to the field.

The general inverse relationship between survival to emergence and fine sediment is valid, but only vertical subsampling of gravel cores from natural environments will show the actual conditions fry must face during emergence.

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Forest Service

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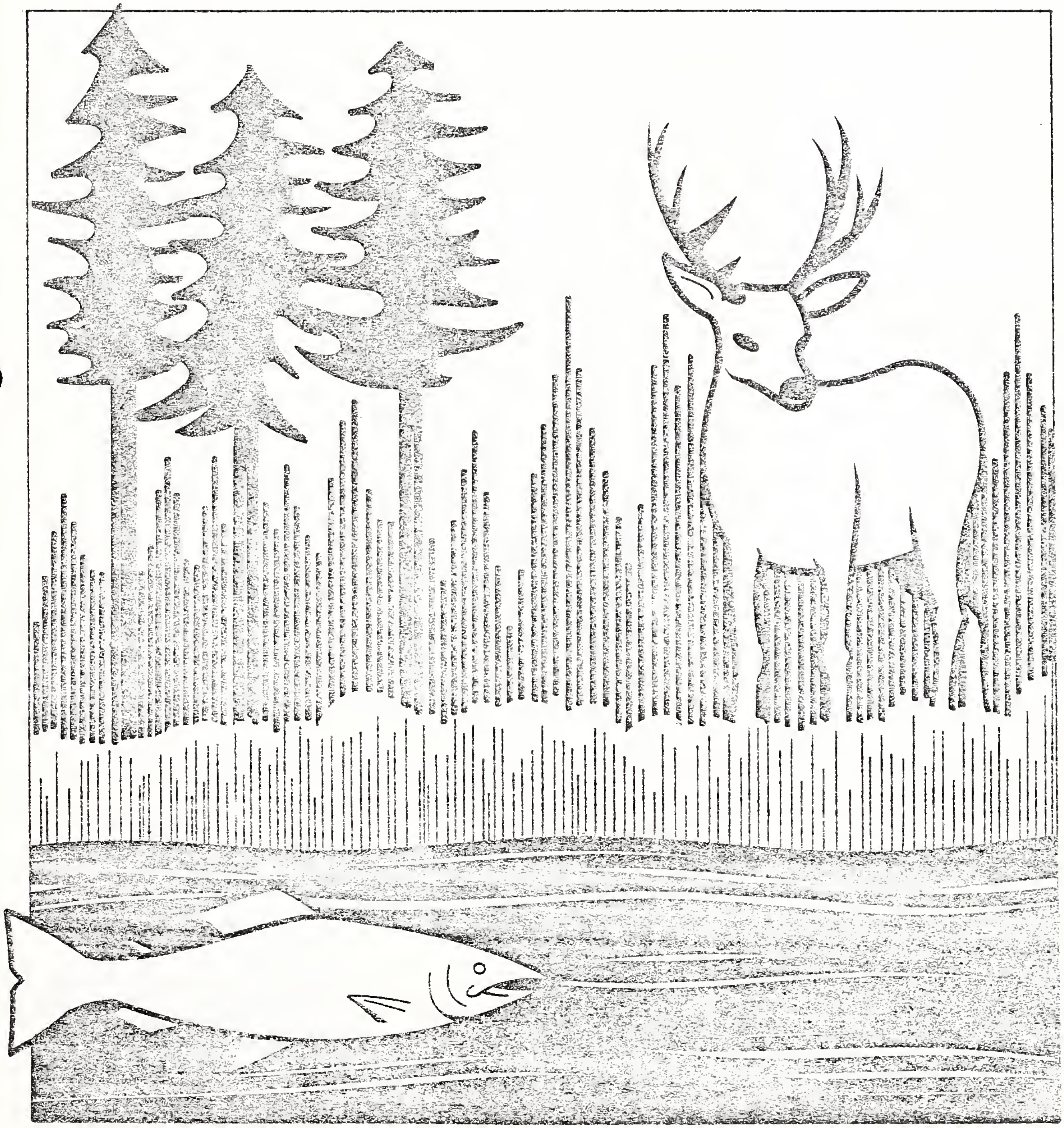
Forest and Range
Experiment Station

General Technical
Report
PNW-146
November 1982

Evaluating Projects for Improving Fish and Wildlife Habitat on National Forests

Fred H. Everest and Daniel R. Talhelm



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Common and Scientific Names

Common name	Scientific name
Coho salmon	*Oncorhynchus kisutch* (Walbaum)
Chinook salmon	*Oncorhynchus tshawytscha* (Walbaum)
Cutthroat trout	*Salmo clarki* Richardson
Rainbow (steelhead) trout	*Salmo gairdneri* Richardson

Abstract

Everest, Fred H.; Talhelm, Daniel R. Evaluating projects for improving fish and wildlife habitat on National Forests. Gen. Tech. Rep. PNW-146. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station; 1982. 12 p.

Recent legislation (P.L. 93-452; P.L. 94-588) has emphasized improvement of fish and wildlife habitat on lands of the National Forest System. A sequential procedure has been developed for screening potential projects to identify those producing the greatest fishery benefits. The procedure — which includes program planning, project planning, and intensive benefit/cost analysis — has nationwide application for both fish and wildlife projects. Fisheries and wildlife values are difficult to assess and available estimates are far from ideal, but better estimates are gradually becoming available.

Keywords: Habitat improvement, wildlife habitat, cost/benefit evaluation, program planning, salmonids.

Contents

Introduction

In the 1970's, Congress formally recognized the potential value of improving fish and wildlife habitat on the National Forests by requiring comprehensive planning for fish and wildlife on National Forest System lands (Sikes Act, P.L. 93-452, as extended in 1974). USDA Forest Service budgets for habitat improvement have grown steadily in recent years, and the National Forest Management Act of 1976 (P.L. 94-588) provided the option to use Knutson-Vandenberg funds for habitat rehabilitation and enhancement within designated timber sales. These recent actions have enabled forest managers to initiate a positive program to improve fish and wildlife habitat on the National Forests. But because more potential projects exist than can be completed annually with existing funds, projects that provide the greatest benefits must be selected.

Biologists, economists, and others who participate in the planning and selection process have difficulty comparing project alternatives objectively because the range of potential projects is so great, and little information has been available for estimating expected benefits from projects. Thus standard, objective evaluation tools, such as benefit/cost analyses, have been difficult to use. Ideally, projects can be compared by evaluating the benefits and costs of each alternative. Project benefits can be defined as the degree to which people are better off with the project than without it. This can be estimated in terms of the aggregate willingness of people to pay to have the project rather than to go without.

Unfortunately, this information is not available for most fish and wildlife projects because of the lack of conventional markets for recreational use of fish and wildlife. How willing recreational users are to pay for changes in the fish and wildlife resource is not generally known. Most economic studies of recreation are not directly usable either because they evaluate something other than willingness to pay for resource use (such as studies of expenditure patterns), or because they estimate willingness to pay for broad, drastic choices (such as the willingness of anglers to pay for a State's entire coastal anadromous fishery rather than not have it at all, called the "all-or-none" value), rather than willingness to pay for the much smaller effects of specific projects (Talhelm 1980). All-or-none values can differ greatly from project values. Even the value of fisheries resources to the commercial fisheries sector is difficult to estimate. The commercial fisheries' net willingness to pay for improved fishing would be the market value of the additional fish minus additional costs of harvesting and bringing them to market. The market values may be well known, but the costs are not.

Biologists working in the National Forests usually cannot obtain the proper kinds of value information for each alternative, so they must use shortcut procedures and rules of thumb instead. Our objective is to explain and illustrate some useful procedures for planning fish and wildlife programs and projects for National Forests, and for estimating fish and wildlife values—and to contrast all of these with ideal procedures. Procedures are illustrated with a specific example from the Siskiyou National Forest of southwest Oregon. Before we discuss procedures for program and project planning, however, we first present a brief explanation of benefit/cost concepts (adapted from Francis et al. 1979).

When we ask whether the benefits of rehabilitation will exceed the costs, what are we really asking? One answer is that we are asking whether, over the long run, the gains outweigh the costs. If so, theoretically, the gainers could pay the capital and labor costs and compensate any losers so that in the end, none would feel worse off and at least one group would feel better off. This is the same question you implicitly ask yourself when you decide whether you should purchase any product in the market. If you voluntarily purchase something at a given price, you do so because you think you will be better off with the product, and your payment will have compensated the producers. Both you and the producers have volunteered to buy and sell, so you both must have decided the exchange was in your best interests. The main difference in rehabilitation is that the beneficiaries of public decisions may not be the only ones who pay the costs, and the losers are not always compensated.

Compensation is considered adequate for our purposes when everyone feels at least no worse off, after the change has been produced and compensation has been paid. Adequate compensation, then, is the minimum required to make people who have given up something feel as well off as before. In other words, it is their willingness to accept compensation. In analyzing public decisions, economists look at two values: the benefits, measured in terms of the maximum the public would be willing to pay; and the costs, measured in terms of the minimum the public would be willing to accept as compensation. These values are easily measured in an ongoing market where payments are offered and accepted, but where is the market for rehabilitation projects?

If long-run benefits exceed long-run costs, the project is probably a good choice for society unless better choices are available because the social gains outweigh the social costs—a gain in aggregate welfare for society. This is because if compensation were paid no one would be worse off and at least some members of society would be better off. These gains and losses can also be assessed in ways other than through economic analysis (such as through political processes or Delphic methods), and sometimes that is necessary if the benefits, costs, or both cannot be accurately estimated. Accurate economic benefit/cost analysis, however, has the advantage of providing confidence that we have arrived at the best decision.

Although benefit/cost analyses assess social choices in an important sense, they do so only in special, confined ways. Certain social values or considerations are usually not addressed in benefit/cost analyses. Those considerations are social equity; human rights or other ethical judgments; economic impacts; and major shifts in social policy.

Social equity in the economic sense is usually thought of as the distribution of income or wealth among members of our society. Because the benefits and costs of projects are measured in terms of willingness to pay or to accept compensation, both of which depend upon the current distribution of wealth, all benefit/costs analyses depend to some extent on current distribution of income. The results might differ if income were distributed differently. In fact, whether or not gainers actually pay losers will have some effect on income distribution, so even this could affect the results of a benefit/cost analysis. In addition, if the existing distribution of income in society were considered unfair, then the results of any benefit/cost analysis might also be considered unfair. The effects of income distribution are ignored in benefit/cost analyses. But political and administrative decisionmaking processes often consider, in one form or another, the equity effects of income distribution.

Human rights and other ethical judgments are also usually ignored in benefit/cost analyses. Obviously these are based on human biases, but benefit/cost analyses do not attempt to distinguish good from bad ethics. Political judgments may, however, and those judgments can override benefit/cost analyses.

To economists, economic impacts are measures of the transfer of income, employment, or both from one region to another or from one sector of the economy to another as a result of some change in the economy. You often hear about dollars spent by recreationists at motels, restaurants, and gas stations, or that commercial fisheries not only employ the fishers themselves but also the processors and perhaps local restaurant personnel. Secondary impacts are usually not relevant in benefit/cost analyses because they are not direct measures of benefits or costs, but of the location of economic activity. The only aspects of economic impacts that should be included in benefit/cost analyses are the positive or negative values of alleviating or causing unemployment of capital, labor, or both, and the value of progress toward or away from any social goal of transferring income and employment from one region or economic sector to another. Communities and regions are often concerned with economic impacts because they greatly influence patterns of local growth and community integrity. These considerations are more likely to form part of a political judgment than a benefit/cost analysis because economists have great difficulty measuring the values represented by such concerns.

Finally, the "macro" judgments of any society are usually considered beyond the useful scope of benefit/cost analysis. For example, the decision to open western North America to homesteading and the decision to put a man on the moon by 1970 both represent deliberate political choices resulting in major changes for society. These kinds of decisions lead to changes that are uncertain or even totally unanticipated at the outset. The decisions are made only because of a strong conviction that the ensuing benefits will more than offset the costs. A traditional, detailed benefit/cost analysis is neither possible nor would it increase public confidence in the merits of this kind of decision. Any major redirection of society or its economy will in itself change the values we rely upon to estimate dollar benefits and costs, thereby reducing the reliability of the analysis.

Social equity and human rights must be considered separately and weighed against the benefit/cost analysis of any project. A logical approach would be to complete the benefit/cost analysis and then to follow any explicit Forest Service guidelines, or if these are unavailable, to ask an advisory committee to rerank projects on the basis of human values. Public hearings could be held or elected representatives might rerank the projects. Obviously, any of these processes might result in socially worse rankings than the original ones, or the costs of the reranking process may be greater than the benefit received. That social equity and human rights values would significantly alter many fish and wildlife project analyses for National Forest projects is unlikely, however.

Program planning—the initial step—is a broad evaluation of the entire range of potential projects, with the objectives of identifying the kinds of projects worthy of more detailed evaluation in the project-planning stage, and understanding how these projects relate to each other and to other goals and objectives of the Forest and of related agencies. Ideally, the benefits and costs of each possible project would be evaluated. In this context, program planning can be viewed as a generalized benefit/cost evaluation, accepting or rejecting whole, broad categories of possible projects to permit the planner to concentrate on the most likely projects.

This process is often not even recognized as a form of benefit/cost analysis. Planners often speak of identifying needs and types of projects that would meet those needs. On closer examination, however, needs are simply results likely to be highly beneficial. An area with few angling opportunities relative to the potential angling effort is spoken of as having a deficit of angler days or a need of so many angler days. In other words, the benefits of providing for additional angler days there are likely to be great. Quantifying potential benefits would be a much more precise way of understanding need, but shortcut procedures are more expedient at this stage.

Identifying Needs

Program development must be responsive to local and national needs for recreation, commercial fishing, or enhancement of habitat for threatened or endangered species. Statewide comprehensive plans, State fish and wildlife plans, and the Endangered Species Act of 1973 (P.L. 93-205) are examples of documents that can be used to identify needs for habitat improvement

Table 1 — Deficits in catch of anadromous and resident salmonids in southwest Oregon in 1980, and Siskiyou National Forest's potential to eliminate deficits by enhancing habitat

Salmonids	Deficit		Enhancement potential
	Number of fish caught	Activity-days	
ANADROMOUS			
Spring chinook	400	3,000	Low
Fall chinook	1,000	6,000	High
Coho	1,000	1,000	Medium
Summer steelhead	400	3,000	Low
Winter steelhead	1,000	5,000	High
Cutthroat	300	300	High
RESIDENT			
Trout in streams	18,000	7,000	Low
Trout in lakes	74,000	30,000	Low

An example from southwest Oregon illustrates how these needs are identified. Data in the Oregon Statewide Comprehensive Plan (USDA Forest Service 1976) indicated that a deficit in supply of most anadromous and resident salmonids occurred within the State in 1980. The most serious deficits in southwest Oregon were associated with fisheries for fall chinook salmon, coho salmon, winter steelhead, and resident trout in lakes and streams (table 1).

Habitat improvement on the Siskiyou National Forest could eliminate a portion of the deficit in recreational angling. The Forest contains portions of 11 river basins producing anadromous salmon and trout; it supports one of the largest anadromous fisheries in the National Forest System. Angling for resident trout in the Siskiyou is minor. The Forest's streams support few fishable populations of resident trout, and only a few small fishing lakes are present. The potential for development or improvement of additional lakes is not great. Clearly, the highest potential for improving fish habitat lies in segments of streams used by anadromous salmonids. Consequently, efforts to improve fish habitat on the Siskiyou were directed first toward eliminating deficits in supply of anadromous salmonids. Improvement of habitat for fall chinook and winter steelhead offers the greatest potential for reduction of deficits.

The capacity of forests in western Oregon to produce resident and anadromous salmonids, as well as other species, varies; regional coordination of proposed projects would in some general way compare expected levels of benefits and costs between Forests. Intraregional coordination of Forest Service projects, and coordination with States and other Federal agencies, should avoid duplication of effort and potential conflicts, as well as direct more funding into the most cost/beneficial areas.

Selecting Projects to Meet Needs

What types of habitat could be improved on the Siskiyou to meet deficits in populations of fall chinook salmon and winter steelhead? To answer this question, factors limiting populations of these species were explored, for both marine and freshwater phases in their life histories. Successful coastwide programs for enhancing hatcheries suggested that the marine environment was not currently limiting any populations of anadromous salmonids produced in southwest Oregon. In fresh water, either spawning or rearing habitat could have been limiting; however, with rare exception, streams on the Siskiyou contained spawning gravels in excess of requirements for seeding the available rearing areas. Increasing spawning gravels would have little or no benefit. Rearing habitat is the limiting factor in most streams on the Forest, primarily because of low streamflows associated with droughty conditions common to southwest Oregon in summer.

What types of projects could most efficiently enhance or expand available rearing areas? Only a few options are available to compensate for shortages of water; a promising one was to allow anadromous salmonids access to unoccupied habitat. Several natural barriers on streams of the Siskiyou block upstream migrants from suitable spawning and rearing habitat. Removing or laddering such barriers could increase production of fall chinook and winter steelhead, and deserved careful study. Program planning started with this type of project but was ultimately expanded to consider all types of projects that could enhance production of anadromous salmonids.

Selecting Geographic Areas for Projects

Other considerations being equal, projects should be located in river basins that have the highest potential to produce additional fish. On the Siskiyou National Forest, the Rogue, Illinois, Chetco, and Elk Rivers probably best fit this category. Projects in those rivers are probably most cost effective—they will probably produce the greatest immediate results (numbers of fish) from funds invested. Potential projects in less productive watersheds, however, should also be considered because the benefits in some, more costly areas could be proportionally much greater than in lower cost areas. Existing data should be compiled for each potential project in which barrier removal or laddering, for example, would benefit anadromous salmonids.

Table 2 — Fish species blocked from upstream access by barriers on Siskiyou National Forest streams

Watershed and streams	Anadromous salmonids: Fall chinook	Coho	Summer steelhead	Winter steelhead	Cutthroat
CHETCO					
Emily	X	X		X	X
Eagle					X
COQUILLE					
South Fork		X		X	
Elk		X		X	
ELK					
Anvil		X	X		X
Rock		X	X		X
ILLINOIS					
Briggs				X	
Cave					X
Collier	X			X	X
Grayback				X	
Indigo	X			X	X
Lawson	X			X	X
Silver	X			X	X
Sucker				X	
ROGUE					
Burned Timber			X	X	
Shasta Costa	X	X		X	X
Stair					X
SIXES					
Dry	X	X		X	X

Selecting Project Sites

Inventory data currently available on the Siskiyou list 18 barriers that are restricting upstream access for anadromous salmonids (table 2), but the inventory is incomplete. Additional stream surveys will increase the number of potential projects. Potential sites should be jointly selected by State and Federal agencies.

General Evaluation of Potential Projects

Each site identified is a potential project, but intrinsic or extrinsic factors might preclude its development. Criteria for eliminating projects might include, at least: the species of fish that will benefit; legal constraints; and administrative restrictions on certain projects or geographic areas. This amounts to a simple form of project evaluation that will probably reduce the number of potential projects. Projects not eliminated should be subjected to project planning to determine their priority for development.

The next step in development of a habitat-improvement program is project planning, in which all barriers listed in table 2 would be subjected to an intense analysis of benefits and costs. A natural falls on Shasta Costa Creek in the Rogue Basin illustrates the process of project planning. All barriers in table 2 should be subjected to the same kind of analysis, and ultimately, so should all other potential projects. The focal point of project planning is a feasibility study, which as a minimum includes: an environmental assessment; an engineering investigation; a preliminary design; and a benefit/cost analysis. Four basic areas of project feasibility will be analyzed during the project-planning process — physical, biological, economic, and social.

The economic analysis is the heart of the feasibility study. All costs associated with the project (planning, construction, operation, and maintenance) must be accurately defined over the expected life of the project. Costs can often be identified precisely, but benefits must also be realistically analyzed in detail — and this is perhaps the most difficult aspect of benefit/cost analysis for projects to improve fish and wildlife habitat.

The first step is estimating increased biological production. How many adult fall chinook salmon or winter steelhead will the Shasta Costa project produce? The question is best answered by relating production per unit area below the barrier to comparable spawning and rearing area above the barrier. Laddering will open about 4.76 km of good spawning and rearing habitat for winter steelhead and cutthroat trout, and about 1.60 km for chinook and coho. Stream surveys and redd counts on Shasta Costa Creek indicate that about 43 pairs of steelhead, 13 pairs of fall chinook, 3 pairs of coho, and 12 pairs of cutthroat spawn annually per kilometer of accessible stream. Applying these data to potentially available habitat above the falls, an increased annual escapement of 200 pairs of steelhead, 20 pairs of chinook, 5 pairs of coho, and 60 pairs of cutthroat could be expected to result from laddering.

Expected catch of salmon is five times escapement, and expected catch of steelhead and cutthroat from Shasta Costa Creek are 25 and 20 percent of escapement, respectively. No benefits would be realized the first 3 years of the project, and only half the potential annual benefits would be realized during the second 3-year period because of the cyclic life-history patterns of these species. These estimates appear conservative, but we think they are realistic based on observed use of downstream waters.

How can the benefits of increased fish production be estimated? Because benefits are measured by the willingness of people to pay for the change, the effects of the project on anglers, commercial fishers, and others must be estimated. Ideally, commercial fishing benefits attributable to the project would be estimated by the resulting increase in commercial fishing revenues (landed value) minus the resulting increase in commercial fishing costs. Because precise estimates of these revenues, and particularly these costs, are usually not available, average revenues and costs may be substituted. These figures are generally available for major commercial species, and average values probably differ little in the long run from values attributable to the project. If the increase in production is great enough to lower prices, the effects on consumers and producers must be considered. Producers benefit because they harvest more fish with the same effort, and consumers benefit from lower prices at the partial expense of producers. The net benefit may be approximated by multiplying the change in price by the average of total production before the change and total production after the change. Detailed econometric studies would be needed to estimate the benefits more precisely.

Angling benefits are estimated by the willingness of anglers to pay for the change. Because no traditional market for angling exists, however, project planners are often at a loss to estimate such values. Furthermore, if a market existed, the values of improvements in angling would vary greatly from site to site, much more like land values vary from place to place than like the relatively uniform values of commercial fish. The value to anglers of a given increase in fish production depends on the relative change in angling quality, the availability of substitutes similar to preproject and postproject angling quality, the availability of substitute kinds of angling, the preferences of anglers for the preproject and postproject kinds of angling, and the accessibility of the site to anglers. One method of estimating the change in angling value is to ask anglers directly how much they are willing to pay for the change. This method is subject to many pitfalls, however, and is not recommended without guidance from experienced researchers. Angler expenditures are not appropriate measures of project values because they measure the cost of angling rather than anglers' willingness to pay for a project. Willingness to pay for a project is an amount in excess of actual expenditures; it can be thought of as an access fee to use the site. For example, in Great Britain, angling rights are privately owned, and from 1973 to 1976 anglers in Scotland paid an average of $175 per fish to rent a section of river for salmon fishing.

Accurate estimates of angling values in the United States are now possible but expensive, requiring highly sophisticated econometric studies of angler travel and expenditure patterns or of anglers' responses to questions about hypothetical situations. An important caution is necessary here. Unless the study is specific to the project site or a site similar in the five respects mentioned in the previous paragraph, the project values will probably differ from the estimated values. Project value can vary that much, even within a

restricted geographic area. In fact, by far most econometric studies of angling values estimate the values of choices that drastically differ from any of the choices usually considered by National Forest planners. Typically the studies estimate the all-or-none value of the fishery investigated — the willingness of anglers to pay to have the present fishery rather than not have it. This is an extreme value, and it is generally higher than most project values because projects on National Forests represent relatively minor changes in the overall fishery. Economists estimate all-or-none values because they are academically interesting and because they represent a clearly identifiable social choice, even if it has practically no direct significance to the projects. More detailed explanations of principles and procedures are available in Clawson and Knetsch (1966), Gregory (1972), and more vigorously in Talhelm (1973), Dwyer et al. (1977), and Freeman (1979).

This leaves the Forest planner with little information on which to estimate project benefits. Even the current values from the Forest and Rangeland Renewable Resources Planning Act used by the Forest Service are based on estimates of all-or-none values. Until better estimates are available, however, project planners have little choice but to follow Forest Service guidelines — as were used for our example, the Shasta Costa project.

Recent National legislation (National Forest Management Act of 1976, P.L. 94-588) requires use of fishery values (and values of other resources) in all land-use plans. The Forest Service, in compliance with P.L. 94-588 and the Forest and Rangeland Renewable Resources Planning Act (RPA), has developed a set of daily consumer benefits (table 3) for use in fishery valuation and economic analysis of habitat-improvement projects (USDA Forest Service 1979).

Table 3 — Net consumer benefits for the USDA Forest Service 1980 Renewable Resources Planning Act Assessment (USDA Forest Service 1979)

Fishery	Consumer benefits/ angler-day	Consumer benefits/ commercial pound
	Dollars	
ANADROMOUS SALMONIDS		
Sport benefits	19.50	—
Commercial benefits	—	0.63
Sport habitat improvement	19.50	—
Commercial habitat improvement	—	0.80
INLAND SPORT FISH		
Cold water/warm water use	5.25	—
Cold water habitat improvement	6.25	—
Warm water habitat improvement	4.25	—

RPA values (table 3) of $19.50 per angler-day for improving habitat for anadromous salmonids and $0.80 per pound for commercially caught salmon were used to estimate consumer benefits for the Shasta Costa project. The procedures are illustrated in figure 1. Most of the predicted net annual benefit of $8,300 is associated with increased production of fall chinook salmon and winter steelhead (table 4). Future evaluations should use the most recent daily consumer benefits recommended by USDA Forest Service. The project will remove about 5.8, 1.5, 10.0, and 8.0 percent, respectively, of the deficit in catch of fall chinook, coho, winter steelhead, and sea-run cutthroat that was expected in southwest Oregon by 1980.

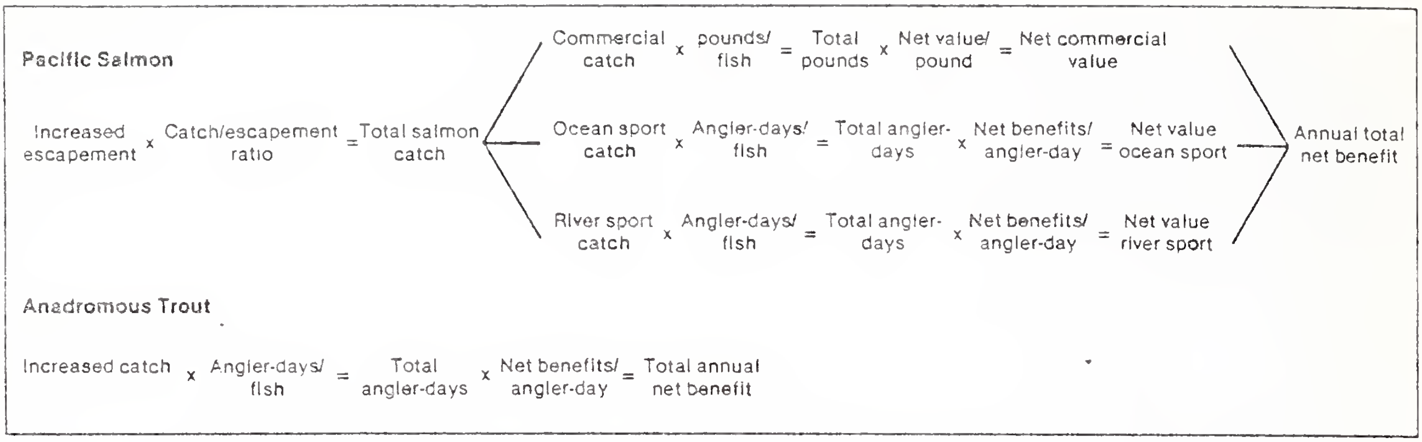

Figure 1. — Procedures for evaluating increased production of Pacific salmon and anadromous trout resulting from habitat improvement.

Table 4 — Expected annual net economic benefits from laddering a natural barrier on Shasta Costa Creek, Siskiyou National Forest

Species	Increased catch	Net consumer benefits, sport	Net consumer benefits, commercial	Total net consumer benefits	"Catch deficit" relieved
	Number	*Dollars*			
Fall chinook	200	1,360	1,210	2,570	5.8
Coho	50	330	190	520	1.5
Winter steelhead	100	4,680	—	4,680	10.0
Cutthroat	24	530	—	530	8.0
Total	374	6,900	1,400	8,300	

Costs associated with the project were also estimated. The Shasta Costa barrier, located 2 km from the nearest access road, is composed of a formation of bedrock and large boulders that creates a cascade about 3.2 m high. A concrete fishway, an aluminum Alaska steeppass, and a bedrock fishway created by blasting were considered possible alternatives for upstream passage of salmonids. Estimated construction costs of a conventional concrete fishway at this remote location exceeded $100,000, and the cost of a steeppass was about $25,000. Steeppasses, however, are easily clogged with debris and need frequent attention during freshets — a major disadvantage in a remote location. A fishway constructed by drilling, blasting, and adding minor supplemental weirs of concrete and reinforcing steel was easy to build at this location, economical, and required

little maintenance. The estimated cost included $930 for planning, $9,170 for construction, and $300 annually for maintenance (table 5). Because anticipated benefits were the same for all three construction techniques, the third alternative was selected.

Once costs and benefits have been estimated, project alternatives may be compared. Costs and benefits anticipated during the effective life of the project (considered to be 20 years in this example) may be listed in a table and discounted back to a common time, usually the year of construction. Discounting is necessary because a dollar today is worth more than the prospect of a dollar at some future date, and discounting determines the present worth of costs and benefits that are incurred or realized in the future. The present worth (discounted value) of a cost incurred in the future is calculated by use of the single-payment, present-worth factor by the formula:

$$P = S \frac{1}{(1 + i)^n},$$

where P = worth of the sum S, n years in the future at interest rate i. For example, the present worth of a $500 benefit expected 2 years in the future at 7-percent interest equals:

$$P = 500 \frac{1}{(1 + 0.07)^2} = \$437.$$

For a more detailed discussion of discounting, see Sheridan (1969) and Grant and Ireson (1970). A planner should use this standard method of discounting and the appropriate current interest rate to estimate present worth of project costs and benefits. In this example, we have used an interest rate of 7 percent as recommended by the Water Resources Council (1973) for water-related development projects (table 6). The benefit/cost ratio (B/C) was derived by dividing discounted benefits by discounted costs. If the ratio is greater than 1, the project is economically sound. The Shasta Costa project, with a ratio of 4.33:1, appeared to be economically sound and highly desirable for development.

Table 5 — Estimated costs of providing fish passage at a natural barrier on Shasta Costa Creek, Siskiyou National Forest

Activity	Cost
PROGRAM PLANNING	*Dollars*
1 person-day	80
PROJECT PLANNING	
11 person-days	750
Travel	100
CONSTRUCTION	
40 person-days	3,500
Travel and per diem	800
Powder	120
Concrete, steel, wood	500
Equipment	3,000
Helicopter time	750
Administration	500
Maintenance, $300 annually	5,700
Total	15,800

If a project B/C ratio is sensitive to changes in interest rates or to errors in estimated costs and benefits, especially if the ratio is near 1:1, the project is considered risky. For example, the Office of Management and Budget (OMB) requires that economic soundness of projects to improve fish habitat considered by the USDA Forest Service be assessed at a 10-percent discount rate. If a slight increase in interest rate changes the B/C ratio from a number greater than one (>1) to a number less than one (<1), the project might be questionable. To test its sensitivity to errors in cost and benefit measurement, the B/C ratio could be recalculated, for example, with the assumption that costs are up 10 percent and benefits are down 10 percent. If slight changes in benefits and costs change the B/C ratio from a number >1 to a number <1, the project might be marginal. Sensitivity analysis is highly recommended because estimates of project benefits are so uncertain.

Table 6 — Discounted benefits and costs of the Shasta Costa project, Siskiyou National Forest, and benefit/cost ratio calculated at a discount rate of 7 percent

Year	Cost	Discounted cost	Benefit	Discounted benefit[1]	Discount factor @ .07
			Dollars		
1	10,100	10,100	0	0	.9346
2	300	280	0	0	.8734
3	300	262	0	0	.8163
4	300	245	4,150	3,166	.7629
5	300	229	4,150	2,959	.7130
6	300	214	4,150	2,765	.6663
7	300	200	8,300	5,168	.6227
8	300	187	8,300	4,831	.5820
9	300	175	8,300	4,514	.5439
10	300	163	8,300	4,219	.5083
11	300	152	8,300	3,943	.4751
12	300	143	8,300	3,685	.4400
13	300	133	8,300	3,445	.4150
14	300	125	8,300	3,219	.3878
15	300	116	8,300	3,008	.3624
16	300	109	8,300	2,811	.3387
17	300	102	8,300	2,628	.3166
18	300	95	8,300	2,456	.2959
19	300	89	8,300	2,295	.2765
20	300	83	8,300	2,115	.2584
Total	15,800	13,202	128,650	57,227	

$$B/C = \frac{57{,}227}{13{,}202} = 4.33$$

[1] Costs are not discounted in year 1, but any benefits realized in year 1 are discounted.

The B/C ratio can be calculated perhaps most efficiently by using the USDA Forest Service, California Region (R-5) computer program, Invest III (USDA Forest Service 1972). The program allows use of three interest rates in determining B/C ratios and has internal provisions for testing sensitivity to changes in anticipated benefits and costs. Invest III sensitivity analysis for the Shasta Costa project indicated that the positive B/C ratio of 4.33 at 7-percent interest is not sensitive to slight changes in interest rate or costs and benefits. If costs are increased 10 percent and benefits reduced 10 percent, the B/C ratio is still a positive 3.55:1. At the OMB-recommended interest rate of 10 percent, the B/C is 3.35:1 and at 6 percent, 4.47:1. Invest III also estimates present net worth at three interest rates and the internal rate of return. The present net worth of the Shasta Costa project are $29,658, $44,057, and $50,253, respectively, at 10-, 7-, and 6-percent interest, and the internal rate of return is 26.6 percent. Figures 2 and 3 list Invest III data for the Shasta Costa project. For detailed information on Invest III, consult R-5 Invest III handbook (USDA Forest Service 1972).

The internal rate of return (IRR) assesses the relative payoff of a project. If a limited budget is available and projects being considered range greatly in size, the easiest way to select the group of projects with the greatest aggregate return within the budget is to compare internal rates of return. Simply rank projects by IRR, and choose the highest set possible within the budget.

Public review is often the final step in determining the soundness of a project. After physical, biological, and economic analyses are complete, an environmental analysis or an environmental impact statement must be written. If a project has a significant impact on the environment, it must have public review. Negative public response could result in a decision to defer or delete the project from further consideration. If the project survives public review, it is ready to be undertaken when funds become available.

For a complete analysis of improvement projects associated with barriers, other potential projects listed in table 2 should be subjected to the planning process. Detailed B/C and IRR analyses of each project should determine priority. Also, because habitat inventory on the Siskiyou is incomplete, the planning process must be kept open to accommodate new information as it is collected.

CARD TYPE	ALT. NO.	BASE NO.	NAME OF FOREST ORGANIZATION							
			NAME OF UNIT							
			NAME OF USER							
			NAME OF PROBLEM							OMB RATE / OPT RATE / RANK RATE
			PROBLEM DESCRIPTION							
			SEQ. NO.	ITEM DESCRIPTION	PMT TYPE	YEAR	YEAR	TOTAL AREA	UNITS	COST OR INCOME DOLLARS

LONG-TERM INVESTMENT ANALYSIS

BENEFIT AND COST EFFECTIVENESS COMPARISON (SEE INVEST III FAST VERSION USER GUIDE)

CARD TYPE	ALT. NO.	BASE NO.	NAME OF FOREST ORGANIZATION							
1		100	SISKIYOU NATIONAL FOREST							
2		100	NAME OF UNIT FISH AND WILDLIFE							
3		100	NAME OF USER EVEREST							
4		100	NAME OF PROBLEM SHASTA COSTA BARRIER 100							OMB RATE / OPT RATE / RANK RATE
5		100	PROBLEM DESCRIPTION SHASTA COSTA FISH PASSAGE FACILITY							1000 / 0700 / 0700
			SEQ. NO.	ITEM DESCRIPTION	PMT TYPE	YEAR	YEAR	TOTAL AREA	UNITS	COST OR INCOME DOLLARS
6			1	1 CONSTRUCTION	01	000	001			-10.100
6			2	1 MAINTENANCE	02	002	020			-300
6			3	1 NET FISH BENEFITS	02	004	006			4,150
6			4	1 NET FISH BENEFITS	02	007	020			8,300
7		100	1							.10
7		100	2							.10
7		100	3							-.10
7		100	4							-.10

LONG-TERM INVESTMENT ANALYSIS

BENEFIT AND COST EFFECTIVENESS COMPARISON (SEE INVEST III FAST VERSION USER GUIDE)

Figure 2. — Standard Invest III (USDA Forest Service 1972) form for benefit/cost analysis. Data are for the Shasta Costa project, Siskiyou National Forest.

Figure 3. — Simulated Invest III output data for benefit/cost analysis of the Shasta Costa project.

Project Evaluation

The final element in program development — perhaps equal in importance to the steps discussed previously — is postdevelopment assessment of the accuracy of the cost and benefit estimates. Long-term projects may be assessed several times over their lives.

The Shasta Costa fishway construction was completed in 1978. Actual construction costs were $1,384 higher than estimated. The annual maintenance fee of $300/year is still anticipated to be correct. Increases in fish production cannot yet be measured, but the fishway is allowing fall chinook salmon and winter steelhead to pass upstream, and projected increases in production will probably occur. Based on the increase in construction costs and the probability that anticipated benefits will be realized, a postconstruction benefit/cost ratio (3.92:1) can be calculated at a 7-percent discount rate. Additional benefit/cost calculations should be made as actual increases in biological production are monitored over the next few years. Results of such evaluations are valuable in planning future projects to enhance habitat and improving precision of future benefit/cost analyses.

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Recent legislation (P.L. 93-452; P.L. 94-588) has emphasized improvement of fish and wildlife habitat on lands of the National Forest System. A sequential procedure has been developed for screening potential projects to identify those producing the greatest fishery benefits. The procedure — which includes program planning, project planning, and intensive benefit/cost analysis — has nationwide application for both fish and wildlife projects. Fisheries and wildlife values are difficult to assess and available estimates are far from ideal, but better estimates are gradually becoming available.

Keywords: Habitat improvement, wildlife habitat, cost/benefit evaluation, program planning, salmonids.

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Riparian Zone and Large Organic Debris

Fred Swanson and Jim Sedell

I. The multiple interactions between forests and streams (fig. 1 Meehan et al. 1977).

A. Overview of interactions

B. Variation in space

C. Variation in successional time

II. Physical conditions and natural and management history of large, woody debris in streams.

A. Functions of woody debris

B. Variation in quantity and spatial distribution

C. Changes through time

1. Natural

2. Managed

III. Forest controls on fish

A. Structural controls

B. Food base controls

C. Temperature control

IV. Management

A. Buffer strips--rotation--salvage

B. Placement of fish habitat improvement structure

1. Where

2. How much

3. How to engineer

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Influences of Riparian Vegetation on Aquatic Ecosystems with Particular Reference to Salmonid Fishes and Their Food Supply[1,2]

William R. Meehan, Frederick J. Swanson, and James R. Sedell[3]

Abstract.--The riparian zone has important influences on the total stream ecosystem including the habitat of salmonids. Shade and organic detritus from the riparian zone control the food base of the stream and large woody debris influences channel morphology. Temporal and spatial changes in the riparian zone, the indirect influences of riparian vegetation on salmonids, and the effects of man's activities are discussed.

INTRODUCTION

Streamside vegetation strongly influences the quality of habitat for anadromous and resident coldwater fishes. Riparian vegetation provides shade, preventing adverse water temperature fluctuations. The roots of trees, shrubs, and herbaceous vegetation stabilize streambanks providing cover in the form of overhanging banks. Streamside vegetation acts as a "filter" to prevent sediment and debris from man's activities from entering the stream. Riparian vegetation also directly controls the food chain of the stream ecosystem by shading the stream and providing organic detritus and insects for the stream organisms.

[1] Contributed paper, Symposium on the Importance, Preservation and Management of the Riparian Habitat, July 9, 1977, Tucson, Arizona.

[2] The work reported in this paper was supported in part by National Science Foundation Grant No. 7602656 to the Coniferous Forest Biome and Grant No. BMS75-07333 to the River Continuum Project. This is contribution No. 283 from the Coniferous Forest Biome and No. 5 from the River Continuum Project.

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WHAT IS RIPARIAN VEGETATION?

Riparian vegetation is at the interface between aquatic and terrestrial environments. It has, therefore, been defined and examined from a number of perspectives. Plant ecologists speak in terms of riparian species and plant communities. The riparian zone may also be defined geographically in terms of topography, soils, and hydrology. We prefer to take a functional approach; that is, to consider riparian vegetation as any extra-aquatic vegetation that directly influences the stream environment.

Consequently, in defining riparian vegetation we must consider the full scope of its biological and physical influences on the stream. Riparian vegetation regulates the energy base of the aquatic ecosystem by shading and supplying plant and animal detritus to the stream. Shading affects both stream temperature and light available to drive primary production; therefore, the balance between autotrophy and heterotrophy is determined by multiple functions of riparian vegetation.

Although imperfect, the stream order system(Leopold et al. 1964) is a useful way to classify elements of a drainage system. In small and intermediate-sized streams (up to about fourth-order) in the Pacific Northwest, riparian vegetation exercises important controls over physical conditions in the stream environment. Rooting by herbaceous and woody vegetation tends to stabilize streambanks, retards erosion, and, in places, creates over-

hanging banks which serve as cover for fish. Above ground woody riparian vegetation is an obstruction to highwater streamflow, sediment and detritus movement, and is a source of large organic debris. Large organic debris in streams (1) controls the routing of sediment and water through the system, (2) defines habitat opportunities by shaping pools, riffles, and depositional sites and by offering cover, and (3) serves as a substrate for biological activity by microbial and invertebrate organisms (Triska and Sedell 1976; Swanson et al. 1976; Sedell and Triska 1977; Anderson et al. in press).

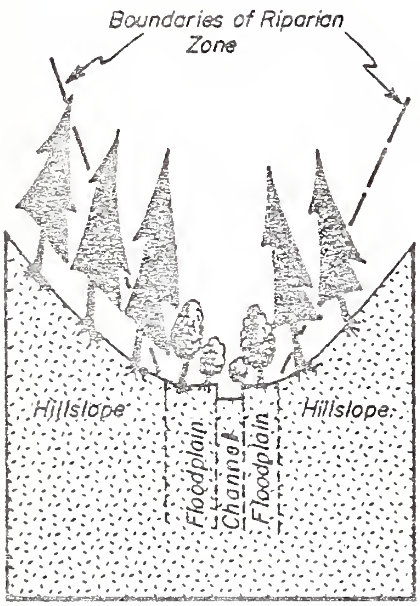
The influences of riparian vegetation on coniferous forest stream ecosystems in the Pacific Northwest are summarized in figure 1. In a functional approach to defining riparian vegetation, all floodplain vegetation as well as trees on hillslope areas which shade the stream or directly contribute coarse or fine detritus to it are considered part of the riparian zone. In the Pacific Northwest, vegetation in the zone of riparian influence includes herbaceous ground cover, understory shrubby vegetation (commonly deciduous), and overstory trees on the flood plain (generally deciduous) and on hillslopes (generally coniferous).

VARIATIONS OF THE RIPARIAN ZONE IN TIME AND SPACE

The character and importance of riparian vegetation varies in time and space. Temporal variation involves patterns of vegetative succession following disturbances. Major processes of vegetation disturbance include wildfire and clearcutting (important to upslope vegetation) and damage due to impact of sediment and floating ice or organic debris during flood flows. Spatial variation occurs along the continuum of increasing stream size from small headwater streams to large rivers.

Temporal Variations of Riparian Zones

The effectiveness of a riparian zone in regulating input of light, dissolved nutrients, and litterfall to the stream varies through time following wildfire, clearcutting, or other disturbances (fig. 2). In the first decade or two following deforestation, streamside vegetation may increase in height growth and biomass more rapidly than upslope communities. Shading of the stream by riparian vegetation gradually diminishes the potential for aquatic primary production until maximum canopy closure. Deciduous shrubs and trees within the riparian zone will contribute most



RIPARIAN VEGETATION

SITE	COMPONENT	FUNCTION
above ground-above channel	canopy & stems	1. Shade-controls temperature & in stream primary production 2. Source of large and fine plant detritus 3. Source of terrestrial insects
in channel	large debris derived from riparian veg.	1. Control routing of water and sediment 2. Shape habitat-pools, riffles, cover 3. Substrate for biological activity
streambanks	roots	1. Increase bank stability 2. Create overhanging banks-cover
floodplain	stems & low lying canopy	1. Retard movement of sediment, water and floated organic debris in flood flows

Figure 1.--Extent of riparian zone and functions of riparian vegetation as they relate to aquatic ecosystems.

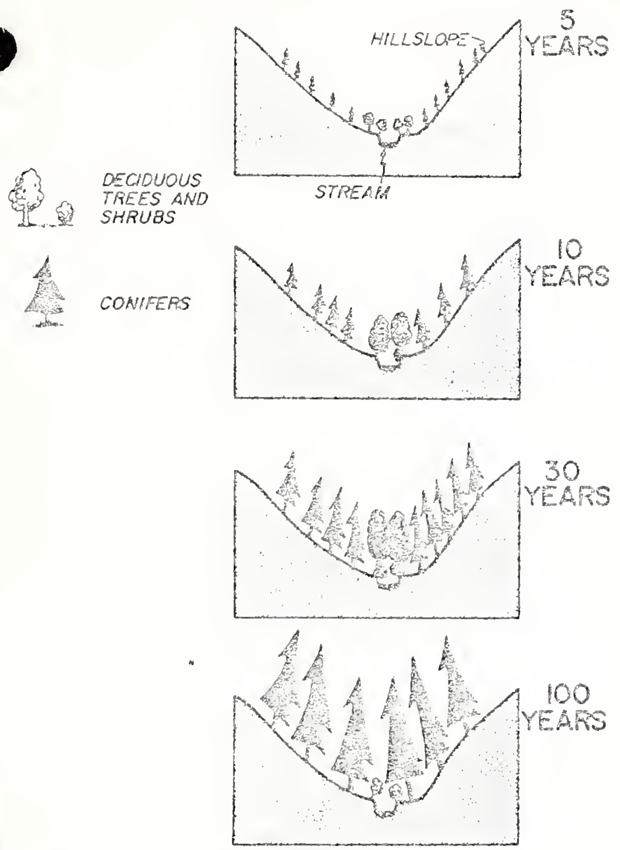


Figure 2.--Changes in the riparian zone through time.

of the litter inputs during early watershed recovery. These deciduous inputs will more readily decompose than coniferous litter which dominates inputs late in watershed recovery and in old-growth forests (Sedell et al. 1975; Triska and Sedell 1976).

The temporal development of riparian zones causes a shift in the energy base of the stream from algae to deciduous leaves to a combination of deciduous and coniferous leaves. The last stage in riparian succession is a complex mosaic of coniferous overstory, deciduous shrub layer, and herbaceous ground cover. Streams flowing through older, stratified forests receive the greatest variation in quality of food for detritus-processing organisms. Herbaceous vegetation is high in nutrient content, low in fiber, and utilizable by stream organisms as soon as it enters the stream. Leaves from the deciduous shrub layer are higher in fiber content and take 60 to 90 days after entering the stream to be utilized fully by stream microbes and insects. The conifer leaves take 180-200 days to be processed. Thus there is a sequencing of utilization of inputs from these three distinctive riparian strata. The results for the stream are rich and diverse populations of aquatic insects which are keyed into the timing and varied quality of the detrital food base.

Spatial Variation of Riparian Zones

A stream should be viewed as a continuum from headwaters to mouth (Vannote, personal communication; Cummins 1975, 1977). The influence and role of riparian vegetation will vary with stream order and position along the continuum. Some broad characteristics of streams and rivers are depicted diagrammatically in figure 3.

Extensive networks of small first to third order streams comprise about 85 percent of the total length of running waters (Leopold et al. 1964). These headwater streams are maximally influenced by riparian vegetation (the ratio of shoreline to stream bottom is highest), both through shading and as the source of organic matter inputs. Even in grasslands, the distribution of trees and shrubs follows perennial and, occasionally, intermittent watercourses except where land use practices have resulted in removal or suppression of riparian vegetation.

These low light, high gradient, constant temperature headwater streams receive significant amounts of coarse particulate matter (CPOM > 1-mm diameter). Their most striking biological features are the paucity of green plant life or primary producers (algae and vascular plants) and the abundance of invertebrates that feed on CPOM (Cummins 1974, 1975). Shredders reduce detritus particle size by feeding on CPOM and producing feces which enter the fine particulate organic matter (FPOM < 1-mm diameter) pool.

Although the transition is gradual and varies with geographical region, the shift from heterotrophy to autotrophy usually occurs in the range of third- to fourth-order streams (fig. 3). Rivers in the range of fourth- to sixth-order are generally wide and the canopy of riparian vegetation does not close over them. Direct inputs of CPOM from the riparian zone are lower in larger rivers because of the reduced ratio of length of bank to area of river bottom.

The importance of floodplain vegetation (mainly deciduous) increases relative to the hillslope species (mainly coniferous) and in a downstream direction. Generally this is so because the floodplain width increases downstream and the canopy opening over larger streams allows greater arboreal expression of deciduous riparian vegetation. Development of deciduous riparian trees is suppressed by shade along small streams.

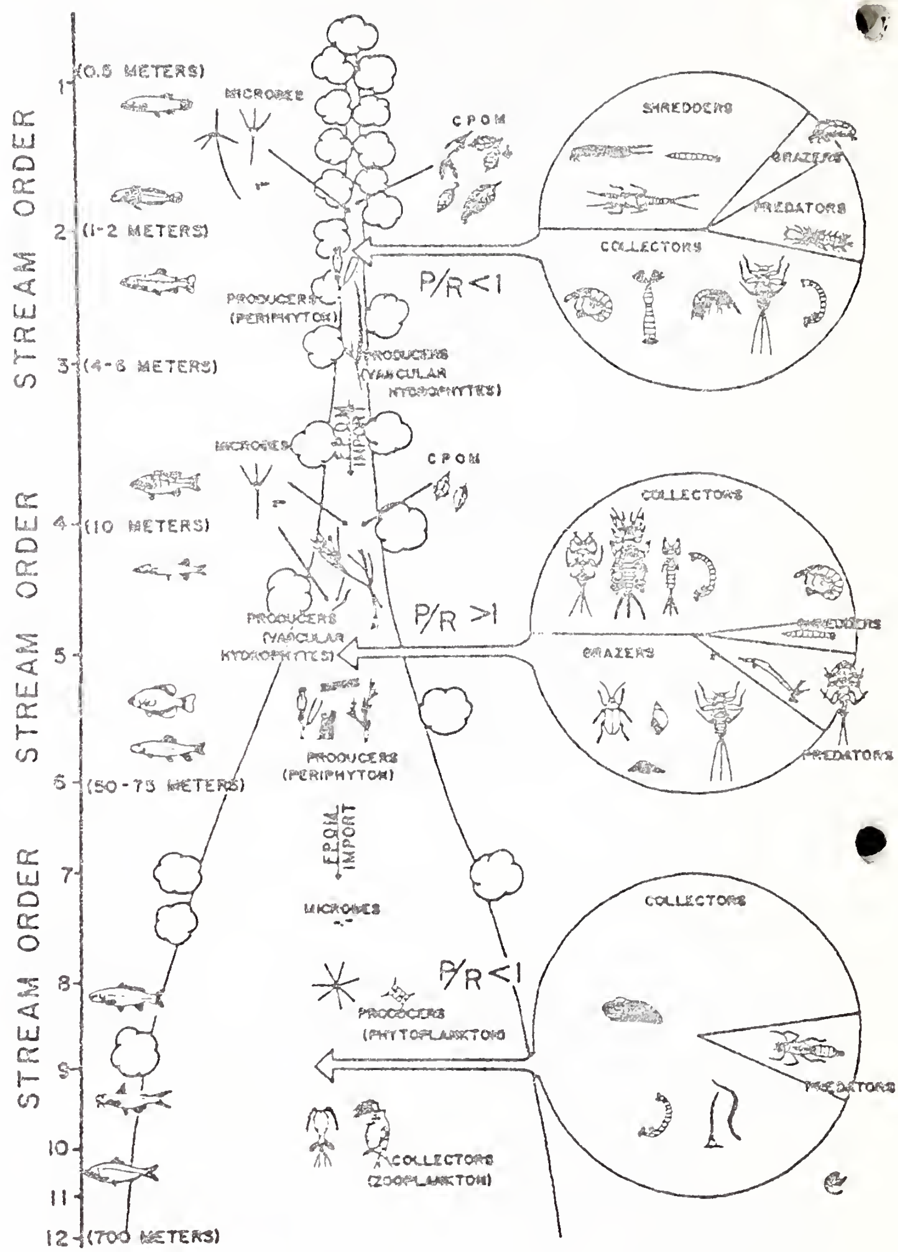


Figure 3.--A diagrammatic representation of some of the changes that occur in running water systems from headwaters to mouth. The organisms pictured are possible representatives of the various functional groups occurring in the size ranges of streams and rivers. Although a large network of smaller tributaries coalesce into larger rivers, the system is shown diagrammatically as a single headwater through all orders to the river mouth (orders and approximate ranges of stream or river width are shown at the left margin). The decreasing direct influence of the adjacent terrestrial vegetation of the watershed and increasing importance of inputs from upstream tributary systems is a basic feature of the conceptual scheme. The proportional diagrams at the right show the changes in relative dominance of invertebrate functional groups from headwaters to mouth. Important shredders include certain species of stoneflies, caddisflies, and craneflies that feed on CPOM (coarse particulate organic matter). Dominant collectors are netspinning caddisflies, blackflies, clams, and certain midge species which filter FPOM (fine particulate organic matter) from the passing water. Also, certain species of mayflies, midges, oligochaetes, and amphipods (may also function as shredders) gather particles from the sediments. Grazers or scrapers include certain species of caddisflies, mayflies, snails, and beetles. In addition to the fish shown at the left, the major predators are helgramites, dragonflies, tanypod midges, and certain species of stoneflies. The midregion of the river system is seen as the major zone of plant growth (algae, or periphyton, and rooted vascular plants) where the ratio of gross primary production (P) to community respiration (R) is greater than 1. Fish populations grade from invertebrate eaters in the headwaters to fish and benthic invertebrate eaters in the midreaches to benthic invertebrate and plankton feeders in the large rivers. (Modified from Cummins 1975).

FOOD BASE AND BIOLOGY OF FORESTED STREAMS

The food base for the biological communities of forest streams consists of leaves, needles, cones, twigs, wood, and bark. The large boles which help shape the small stream are usually biologically processed in place. The input of bole material to the stream is not a regular annual occurrence. Leaves, cones, twigs, lichens, and other components of fine litter have a reasonably predictable timing of input to and export from streams. Of the organic material which falls or slides into first-order streams every year, only 18-35 percent may be flushed downstream to higher order streams. These streams are very retentive, not mere conduits exporting materials quickly to the sea. Sixty to 70 percent of the annual organic inputs are retained long enough to be biologically utilized by stream organisms. Big wood debris dams serve as effective retention devices for fine organic material, allowing time for microbial colonization and insect consumption of this material. Functionally the invertebrates of streams flowing through forests have evolved to gouge, shred, and scrape wood and leaves and to gather the fine organic matter derived from breakdown of coarser material (Cummins 1974; Anderson et al. in press).

Woody debris and leaves, the two major allochthonous components entering a stream from the riparian zone, operate in different ways in relation to quantity, quality, and turnover time of standing crop. The leaves form a small pool of readily available organic material, while the wood forms a large pool of less available organic matter. The slowly processed wood also constitutes a long term reserve of essential nutrients and energy. The composition, metabolic structure, and nutrient turnover time of the particulate organic pool effectively provide both flexibility and stability within the system.

The amount of debris processed in a defined reach of stream depends on two factors: (1) the nature of the debris (abundance and species of wood or leaves) and (2) the capacity of the stream to retain finely divided debris for the period of time required to complete processing. Debris undergoing utilization by stream biota may either be utilized fully within a stream reach or be exported to a downstream reach. Processing continues as small debris moves along the drainage because export from one reach constitutes downstream input. Processing includes both material used metabolically by bacteria and fungi and those debris pieces physically abraded by mineral sediment or by insect consumption. In all cases, the debris is broken into smaller pieces which increases the surface- to-volume ratio and makes a debris particle increasingly susceptible to microbial attack.

Wood in streams is a substrate for biological activity and it creates other habitat opportunities by regulating the movement of water and sediment. To measure the importance of large organic debris from the riparian zone in streams, Swanson and Lienkaemper (unpublished data) examined several streams and measured percent of stream area in (1) wood, (2) wood-created habitat, principally depositional pools, and (3) nonwood habitat such as bedrock and boulder cascades. In a 245-m section of Mack Creek, a third-order stream flowing through an old-growth Douglas-fir stand in the western Cascade Range, Oregon, 11 percent of the stream area is in wood, 16 percent in wood-created habitat, and 73 percent in nonwood habitat. Figure 4 shows an example of the distribution and quantity of debris in a section of Mack Creek. In a first-order tributary draining 10 ha, wood comprises 25 percent of the stream area and another 21 percent is habitat created by wood. Much of the biological activity by detritus-processing and consumer organisms is concentrated in the areas of wood and wood-created habitat. Each habitat type has a different faunal composition.

Wood Habitat Community

Wood habitat communities are distinctive. The primary utilizers are beetles, midges, and snails. In addition to the food supplied to the major wood eaters, the surface area and large number of protective niches on wood afford considerable living space and concealment. Wood is used for oviposition, as a nursery area for early instars, for resting, molting, pupation, and emergence. Because of its unique capillary properties, it affords an ideal air-water interface where gradients of temperature and moisture can be selected by different taxa for various activities.

Wood-Created Habitat

The depositional areas behind large debris are prime areas for processing leaf material and the fine organic matter derived from wood. These areas are richer than the wood habitat community both in numbers and biomass of invertebrates. Leaves and the shredders (primarily caddis- and craneflies) are concentrated in these areas. Many of the shredders feeding here will use the wood habitat to molt, pupate, and emerge.

The difference in invertebrate biomass on leaves and wood is attributed primarily to differences in food quality. Although both are low in nitrogen compared with periphyton, seeds, or fresh macrophytes, the wood is so high in the refractory components lignin and cellulose that it becomes available at a very slow rate. The greater surface area and penetrability of leaves results in microbial con-

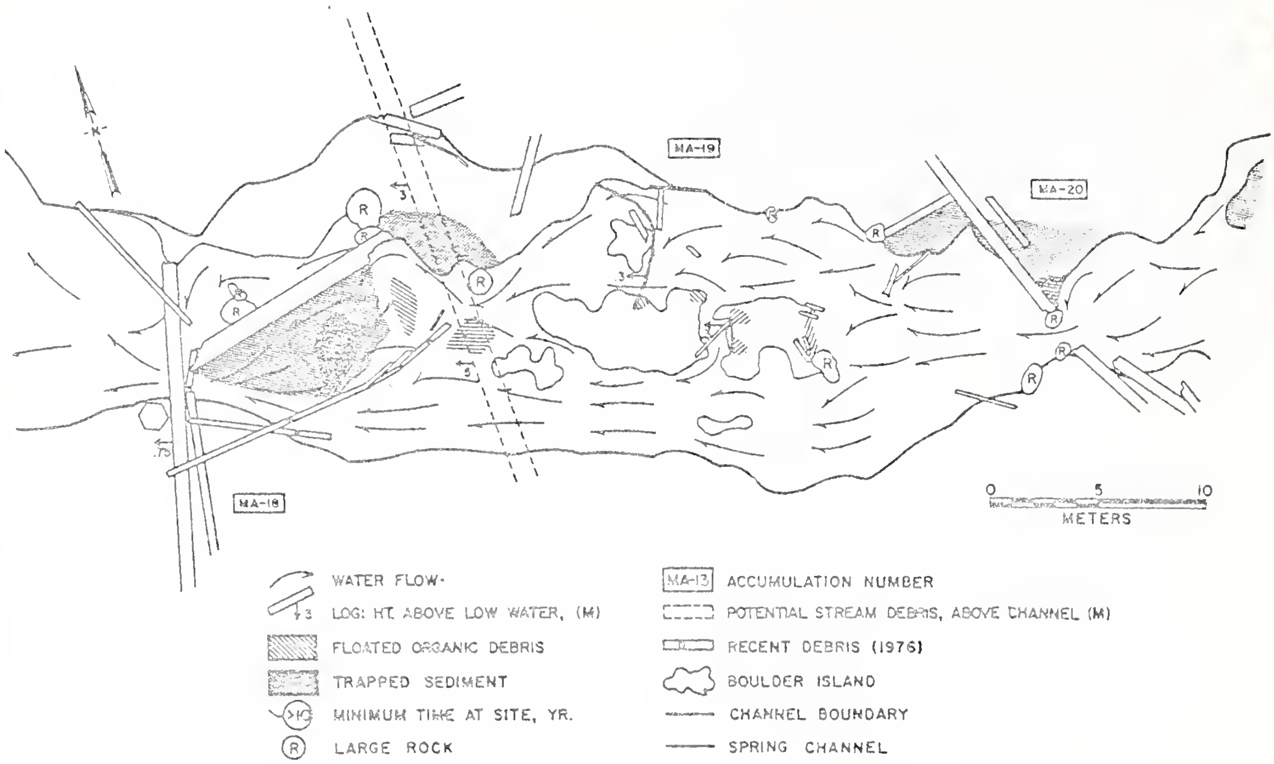


Figure 4.--Distribution of debris in a section of Mack Creek, western Oregon. Courtesy of George W. Lienkaemper.

ditioning occurring within months, compared with years for wood. Conditioning is a key factor in the debris becoming available as food for the invertebrates.

RELATIONSHIP OF RIPARIAN VEGETATION TO SALMONIDS

Direct Influences

The previous discussion has described how riparian vegetation contributes to primary stream productivity through input of organic material and nutrients which are utilized by various components of the stream biota. These relationships directly affect the production of fish by establishing the basic components of the food chain which eventually lead to the fish themselves. Likewise, necessary portions of salmonid habitat are created by large pieces of debris from the riparian zone. Logs and debris jams create pools and protective cover. This type of habitat also provides communities of benthic organisms different from those associated with the shallower and faster waters of riffles and runs. This increase in diversity of invertebrates provides a more useable food base for the fishes, which depend to a great extent upon them. A large part of the diet of fish in the family Salmonidae (the various Pacific salmon, trout, and char) is aquatic insects and other invertebrate organisms.

Indirect Influences

In addition to the effect of riparian zone material which directly becomes a part of the stream system, streamside vegetation has many important indirect influences on the habitat of salmonids.

Water Temperature

The principal source of heat which raises water temperatures is direct solar radiation (Brown 1969). Consequently, streamside vegetation is important in maintaining water temperatures suitable for spawning, egg and fry incubation, and rearing of anadromous and resident salmonids. Several studies in the last decade have demonstrated how streamside vegetation directly controls water temperature (Levno and Rothacher 1967, Brown and Krygier 1970, Meehan 1970, Burns 1972). The literature is also rich with documentation of the effects of streamside canopy removal on stream temperatures (Hall and Lantz 1969, Meehan et al. 1969, Brown and Krygier 1970, Burns 1972, Moring 1975).

Stream temperature is directly proportional to surface area and solar energy input, and inversely proportional to streamflow (Gibbons and Salo 1973). Therefore, small forested streams are the most susceptible to temperature

change. The insulating effect of riparian vegetation is thus of primary importance in maintaining acceptable stream temperatures in the many small streams which cumulatively produce a significant portion of the salmon and trout populations of the Western United States.

Sediment

Another major function of riparian vegetation is to act as a buffer or "filter" against sediment and debris which would otherwise be deposited in the stream. Surface runoff is a primary vehicle for the transportation of sediment to streams from adjacent sources, either natural or man-created. The herbaceous communities within the riparian zone are effective in reducing the impacts of this runoff, and the larger shrubs and trees prevent larger debris from entering the stream channel. The value of streamside vegetation for stream protection has been quantified in economic terms by Everest (1975).

Sediment which affects salmonids occurs in two general forms. As suspended sediment, it can be harmful if concentrations are high and persistent (Cordone and Kelley 1961). Under these conditions, silt may accumulate on the gill filaments and actually inhibit the ability of the gills to aerate the blood, eventually causing death by anoxemia and carbon dioxide retention.

Bedload sediment, however, probably limits salmonid production more than suspended sediment. Excessive deposited sediment reduces the flow of intragravel water, which in turn limits the supply of oxygen available to incubating eggs and alevins, and hinders the removal of metabolic waste products (Sheridan 1962, Vaux 1962, Cooper 1965, McNeil 1966). Bedload sediment may also act as a physical barrier, preventing the emergence of newly hatched fry up through the gravel (Koski 1966, Hall and Lantz 1969).

Another effect of sediment is the alteration of habitat used by aquatic insects (Wagner 1959) which directly relates to the growth and condition of the fish which utilize them. Although biomass may not decrease, the species composition may change such that the new forms are not as readily available to the fish.

Cover

The extensive rooting of herbaceous riparian vegetation aids in streambank stabilization. As a result, where streamside vegetation is intact, the occurrence of undercut banks is higher. This is prime habitat for trout and young salmon. Overhanging streamside vegetation also acts as escape cover and in some instances as a deterrent against predation by birds and mammals.

Insects

As discussed earlier, riparian vegetation contributes to the food base of stream biological communities in the form of wood and other organic debris. In addition, streamside vegetation is important in directly providing insects to the stream which then become part of the available fish food. Terrestrial insects which are associated with the various strata of the riparian zone become "accidental" fish food items. Many of the aquatic insects use streamside vegetation during emergence and in the adult stages of their life cycle.

EFFECTS OF LAND USE PRACTICES

Many of man's activities affect the riparian zone to varying degrees. We must consider logging and road construction to be among the most severe disturbances. Until recently it was common practice to clearcut timber to the stream's edge. In addition to removing the trees which provided shade to the stream surface, the understory vegetation and ground cover were usually cut down or severely disturbed. In recent years, the importance of the smaller streams has been more fully recognized and buffer strips along streams are often left.

The riparian zone is also affected by livestock grazing. In addition to cropping off much of the herbaceous vegetation along streambanks, livestock also use the smaller shrubs and young trees as forage. As a result, much of the ground cover and many of the plants which provide shade to small streams are removed. The soil along the streams is compacted by trampling, and together with the removal of the "filtering" plants a situation is created which promotes the addition of fine sediment to the streams. Wild ungulates also utilize the riparian zone, but their presence is much less noticeable than that of cattle and sheep. A workshop was conducted in Reno in May 1977 to bring together existing knowledge on the relationships between livestock and fisheries, wildlife, and range resources. A large part of the material which was discussed at this workshop concerned the riparian zone, and will soon be available.[4]

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[4] USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, California (in press).

SUMMARY

The riparian zone is a very important area influencing the habitat of salmonids. Much of the wood which forms the food base for stream biota comes from the riparian zone. This same wood, when it falls or slides into a stream, has an important role in shaping the stream and creating its habitat types. Streamside vegetation provides shade to the stream surface, thereby maintaining water temperatures acceptable to salmonid fishes. The roots of woody and herbaceous plants provide streambank stability and help to create overhanging banks, an important component of salmonid habitat. Streamside vegetation provides habitat for the later life history stages of aquatic insects and for terrestrial insects which accidentally become part of the food utilized by salmonids.

When the riparian zone is affected by man's activities, the quality of fish habitat will likewise be affected.

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The Multiple Linkages of Forests to Streams

Kenneth W. Cummins

INTRODUCTION

Headwater stream ecosystems in forested watersheds are intimately related to their vegetative setting. The riparian zone, the source area of soil-forest products which enter the stream, contributes to channel stability and generates biologically active organic substrates. Large woody debris often constitutes stable geomorphic features which retain mineral sediment and finer organic material (Swanson and Lienkaemper, 1978; Swanson, Triska, this volume). Inputs of organic solutions and particulates (and inorganic nutrients) provide energy for the stream community over an annual cycle (Fisher and Likens, 1973). The stream also represents a potential source area for the riparian zone of the forest ecosystem at times of greater than bankfull discharge (Merritt and Lawson, 1979).

Thus, because of the interactive nature of forest-stream ecosystems, stream community structure and function should be studied within a watershed context (Cummins, 1974; Hynes, 1975).

BACKGROUND

Prior to the 1960's the primary emphasis in stream ecosystem investigations was on invertebrates as food organisms for game fish in specific stream reaches. In these studies a wide variety of methods was employed to sample plants and animals associated with the channel sediments (see review by Cummins, 1962), but the key role played by the watershed in supplying organic substrates utilized by stream organisms was largely neglected.

The heterotrophic nature of forested, headwater stream ecosystems and their allochthonous-based energy source was formally recognized in the early 1960's (Hynes, 1963; Ross, 1963). A major development of the 1970's has been the measurement of watershed material-balance budgets. Such studies have shown that streams are not merely conduits that export forest ecosystem products from within the boundaries of surface watersheds and subsurface source areas, but rather that they store and biologically process organic inputs (Fisher and Likens, 1973; Sedell et al., 1974). Budgets for reaches, rather than for entire watersheds, require that appropriate segments be chosen for study with all inputs adequately taken into account (Fisher, 1977). In all budget studies it is important to measure storage carefully and to relate release from (or accrual to) storage to the annual inputs. Also, the losses to and introduction from storage must be related to the seasonal and long-term flow regime (Swanson, this volume).

PERSPECTIVES

The 1970's also have involved the development of conceptual models of headwater stream ecosystem structure and function (Cummins, 1974; Minshall, 1978). Refinement of existing models and elaboration of new ones will undoubtedly continue to be a major feature of stream-watershed research in the 1980's. Examples would be evaluating and testing the "River Continuum" and "Nutrient Spiraling" hypotheses. The former depicts stream-river drainage nets as continua of biological organization that reflect geomorphic control (Cummins, 1975; Vannote et al., 1980) from low order headwater streams to higher order receiving rivers (Strahler, 1957; Leopold et al., 1964). "Nutrient Spiraling" (Webster, 1975) refers to the partially open nutrient cycles characteristic of running waters. Portions of the inputs to a given reach are stored and processed and some fraction released

downstream. The incomplete efficiency of storage and processing provides energy and inorganic nutrients for downstream communities. The more efficient reaches (i.e., higher retention and processing) are considered to have the "tightest" spirals.

The River Continuum

A major distinction between lotic ecosystems can be made on the basis of the relative importance of in-stream primary production versus inputs of terrestrial origin as the major source of organic matter for community processes (Vannote et al., 1980). In forested ecosystems, small, shaded, cool headwater steams (approximately orders 1-3) may derive more than 90 percent of their organic carbon from the terrestrial surroundings (Fisher and Likens, 1973; Sedell et al., 1974). The riparian zone vegetation functions both in light attenuation and as the source of allochthonous inputs, including long-term structural (wood debris) and annual energy supplies.

The ratio of daily gross primary production (P) to total daily community respiration (R) (Odum, 1956) reflects the relative dominance of autotrophy versus heterotrophy. However, as Minshall (1978) has shown, even when primary production exceeds upstream and riparian inputs of organic matter, the in-stream derived organic substance is used primarily in a moribund state in detrital food chains. Where riparian vegetation has been removed, as in clearcut timber harvest, or is naturally sparse (high altitudes and latitudes and xeric regions), autotrophy dominates ($P/R > 1$). In wide shallow, generally warmer, well-lighted midsized rivers (orders 4-6), primary production is also the dominant source of organics.

In addition to increases in primary production related to higher light regimes, another significant feature of the adjustment of biological communities to changes in geomorphology, channel configuration, and vegetational setting downriver (along the "continuum") concerns the size distribution of the particulate organic matter (POM, > 0.5 μm particle size) resources. Headwater streams characteristically have greater inputs of coarse material (CPOM, > 1 mm particle size) and, therefore, greater concentrations of the microbial and macrobial biota for which coarse material is the primary nutritional resource (Cummins, 1974). With increasing stream size and reduced importance of direct riparian inputs, a larger proportion of the POM is fine particulate organic matter (FPOM, < 1 mm particle size) transported from the headwater drainage net. The greater abundance of FPOM is reflected by a change in community structure; for example, larger populations of collectors (filter feeding invertebrates) (Wallace and Merritt, 1980).

Nutrient Recyling

Present research on nutrient relationships--particulate and dissolved (DOM, < 0.5 μm) organic matter and inorganic ions--viewed as partially closed cycles, points up the need for measurements of both physical storage and biological processing. The use of radioactive tracers (Ball et al., 1963; Ball and Hooper, 1963) or stable isotope ratios--e.g., $^{13}C/^{12}C$--(Rau, 1978) can provide data for determining pathways and residence times of nutrients in stream ecosystems.

Storage pools or compartments can be defined as locations where organic matter accumulates and is processed (utilized) at rates slower than the average or exposed (oxygenated) sites in the channel. There are three general areas: the deep sediments (low oxygen), the inner core of woody debris jams (low oxygen), and the upper bank or floodplain (low moisture). When organic material buried in the sediments and within debris jams is excavated or that on the upper bank is captured, and re-enters the aerobic stream channel processing regime, it is utilized at a faster rate (Cummins and Klug, 1979; Merritt and Lawson, 1979). Thus, the annual--and longer--hydrographic pattern is critical in determining the proportion and timing of processing and export of annual terrestrial inputs.

Along with channel and upper bank storage or retention, biological processing is the major control of quantities of material introduced and their rates of recycling. The prediction from the "River Continuum" hypothesis (Vannote et al., 1980) is that spiraling would be tighter, especially for coarse particulate organic matter (CPOM, > 1 mm particulate size), in headwater streams due to more efficient retention and processing.

ORGANIC RESOURCES AND FUNCTIONAL GROUPS

The quantities and qualities of organic resources exert a major influence on stream community structure (Cummins, 1974; Hynes 1975; Minshall, 1978) which is expressed the functional roles of macroinvertebrate species. Different functional groups have adapted morphologically, behaviorally, and

Table 1. Categorization of organic resources in lotic ecosystems (modified from Cummins and Klug, 1979)

Resource category	Approximate particle size range	Major sources	Ratio of carbon to nitrogen (C/N)	Macroinvertebrate functional feeding group using resource
Periphyton (microproducers)	< 500 > 10 μm	In-stream photosynthesis	5-10:1	Scrapers
Macrophytes (macroproducers)	> 1 cm (some macroalgae) < 1 cm > mm	In-stream photosynthesis	13-70:1	Shredders, scrapers
Woody detritus	> 10 cm (coarse) < 10 cm > 10 mm (fine)	Riparian zone (upstream tributaries during floods)	200-1,300:1	Shredders (gaugers)
Nonwoody detritus (particulate organic matter or POM)	> 0.5 μm	Riparian zone, upstream	70-80:1; (microbial portion 10-11:1)	
Coarse (CPOM)	> 1 mm	Riparian zone	20-80:1	Shredders
Fine (FPOM)	< 1 mm > 0.5 μm	Upstream, riparian zone	7-40:1[1]	Collectors
Dissolved organic matter (DOM)	< 0.5 μm	Subsurface source areas, upstream, riparian zone	< 17 (labile portion lower)	None
Animal tissue	> 100 μm (microforms > 10 m)	In-stream	< 17	Predators

[1] A significant portion of the nitrogen may be biologically very resistant.

physiologically to utilize various components of the spectrum of available resources (Cummins, 1974, 1975; Merritt and Cummins, 1978; Cummins and Klug, 1979).

Organic Resources

The basic categories of organic resources in running waters (Table 1) differ significantly in nutritional content as defined by microbial and animal growth. In addition to animal tissue used by predators, there are three general classes of organic resources: (1) those with chlorophyll (living micro- and macroproducers), (2) detritus, ranging in size from large wood to particles less than 1 μm and all with associated microorganisms, and (3) dissolved organics (which can be taken up by microbes). If the ratio of carbon to nitrogen (C/N) is used as an index of resource nutritive value, ratios of 17 or less are generally considered in the high quality range (Russell-Hunter, 1970). However, low ratios may be misleading, as in the case of some FPOM (Table 1), because the nitrogen may be in a recalcitrant form (Ward and Cummins, 1979).

Fungi are relatively more important on CPOM where mycelia can develop, and bacteria are predominant on FPOM (Cummins and Klug, 1979). Because the microbial biomass associated with detritus is nutritionally superior (e.g., low C/N) to the organic particle substrate which is high in cellulose and lignin, it exerts the major control on the rate of detritus processing. This is mediated both through direct microbial metabolism of the detrital substrate and regulation of invertebrate feeding (Petersen and Cummins, 1974). Substrates, such as different species of leaf litter, vary in the rate at which microbial colonization and metabolism and, therefore, invertebrate feeding proceed. Thus, differences in quality of inputs are realized as differences in stream community metabolism.

The distribution of detrital size fractions in stream ecosystems is a function of the vegetative and soil characteristics of the riparian zone, hydrologic events, and biotic processing. Dissolved organic matter (DOM) generally accounts for 50 percent or more of the total annual organic flux in forested headwater streams (Fisher and Likens, 1973; Sedell et al., 1974). A significant proportion of the DOM generated is quite labile, being physically adsorbed and flocculated, and biologically incorporated by microorganisms at rates approximately equal to its production. This is exemplified by similar measured daily changes in DOM as compared to those observed annually (Cummins et al., 1972; Manny and Wetzel, 1973). The rapid incorporation of the labile fraction of DOM onto particles and into microbes constitutes the important retention characteristic of streams because of the reduced probability of export of particles as opposed to solutions. Of the remaining annual organic flux, about one-half is fine particulate organic matter (FPOM); the greatest percent of CPOM is found in headwater streams, reflecting the close association with the riparian zone.

Annual POM input, exclusive of large woody debris, to headwater forested streams ranges from 300 to 800 g AFDW m^{-2} (Anderson and Sedell, 1979). Although annual inputs may be low, headwater streams characteristically have large standing stocks of large wood (approximately > 2 cm): from 1 to 2 kg m^{-2} in Michigan streams to 10 to 15 kg m^{-2} in western Oregon streams (Anderson et al., 1978; Swanson and Lienkaemper, 1978). The coarse woody debris undoubtedly plays a major role in retaining nonwoody POM inputs, resulting in mean annual standing stocks of approximately 200 to 500 g AFDW m^{-2}.

Macroinvertebrate Functional Feeding Groups

Recognition of stream microinvertebrate functional groups (Fig. 1) has shown considerable promise as a tool for assessing the ecological state of a running water community (Cummins, 1974; Merritt and Cummins, 1978). The relative abundances of the groups reflect environmental conditions, particularly the quantity and quality (i.e., nutritional value) of particulate organic matter inputs and periphyton growth. Arduous and incomplete efforts at taxonomic description can be reduced or circumvented by concentrating on morphological-behavioral adaptions for food acquisition. In addition, because most species are omnivores, this method avoids the lack of resolution associated with concentration on macroinvertebrate diets. Thus, the ratios of various functional groups reflect the nature of the organic food resources available (Cummins and Klug, 1979; Wiggins and Mackay, 1979).

There are five basic macroinvertebrate functional feeding groups. Figure 1 links each group with a nutritional resource that it is morphologically and behaviorally adapted to harvest and physiologically adapted to assimilate. The highest quality nutritional resources are animal tissue, nonfilamentous periphytic algae, and the microbial biomass component of detritus (Table 1) (Anderson and Cummins, 1979).

The CPOM: fungal-bacterial:shredder association (Fig. 1), is exemplified by large invertebrates such as larvae of the cranefly Tipula, which feed on conditioned leaf litter. Conditioning involves rapid leaching of soluble organics followed by colonization and growth of aquatic fungi and bacteria. After microbial populations have softened the substrate, shredders begin actively feeding on CPOM (Cummins, 1974; Cummins and Klug, 1979). Shredders selectively feed on the CPOM with the maximum microbial biomass, and account for at least 30 percent of the total processing (conversion of CPOM to CO_2, FPOM, and consumer biomass)(Petersen and Cummins, 1974). The shredder functional group represents the closest invertebrate linkage with the riparian zone, with growth and survival dependent upon the quantity and quality of the terrestrial inputs.

The FPOM:bacterial:collector association (Fig. 1) includes macroinvertebrates that feed by filtering particles from the passing water, for example, with filtering fans (blackflies) or silt nets (net-spinning caddisflies), and those that gather particles from the stream bottom sediments (many species of midges). Although collectors require the presence of microbial biomass on ingested FPOM for adequate nutrition, they show less adaptation for selective feeding (i.e., selection for highest food quality) than shredders (Cummins and Klug, 1979). The relationship of collectors to the riparian zone is less direct because a significant portion of the FPOM is generated within the stream ecosystem (Fig. 1). Therefore, the ratio of shredders to collectors in a stream community reflects the balance between CPOM and FPOM and the relative dominance of the riparian zone.

Macroinvertebrates of the periphyton: scraper association have adaptations for removing attached algae (primarily nonfilamentous forms) from surfaces (Fig. 1). Because they frequently feed in exposed sites, scrapers are also adapted morphobehaviorally for maintaining position in the current; for example, the heavy mineral cases of scraper caddisflies or the dorsoventral flattening of heptageniid mayflies that allows them to avoid the main force of the flow. Abundance and growth of scrapers

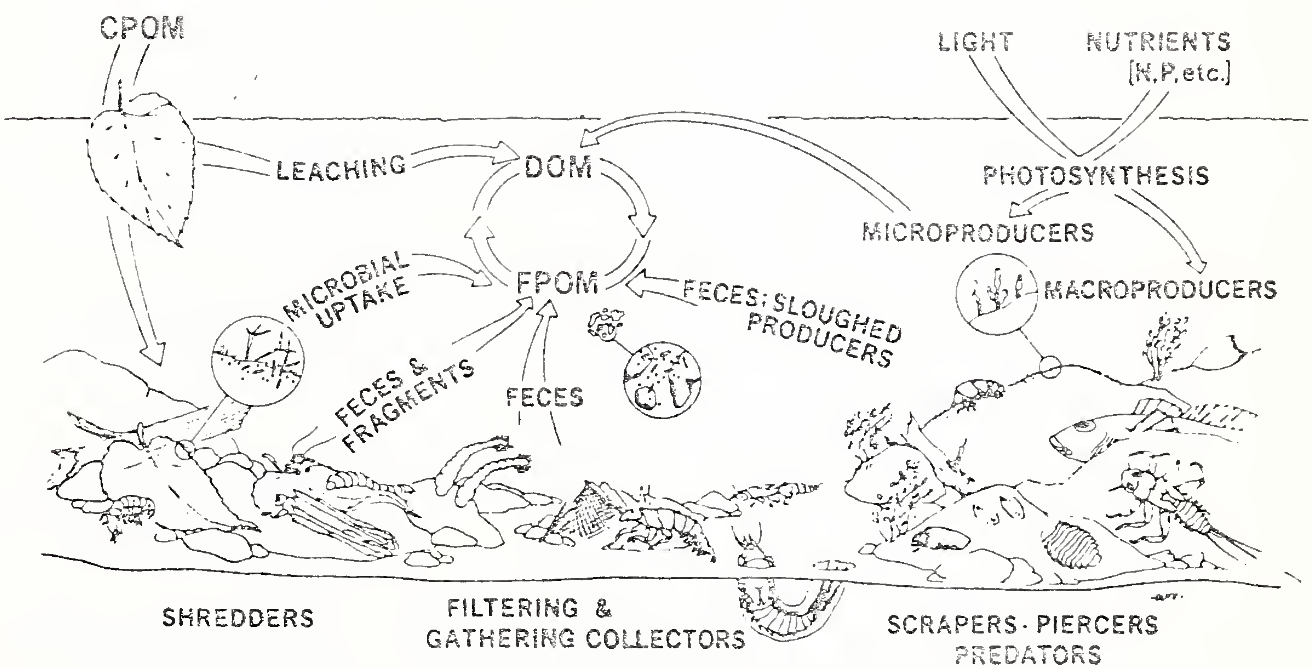

Figure 1. Diagrammatic representation of major resource inputs and partitioning among invertebrate functional groups in forested, headwater stream ecosystems. The major inputs shown, CPOM, light, and nutrients (FPOM and DOM also enter from the riparian zone, not shown), are partitioned among five general processing subsystems associated with macroinvertebrate functional feeding groups. These are the CPOM:fungal-bacterial:shredder; FPOM:bacterial:collector; algal:scraper; macrophyte:piercer; and predator:prey associations. Production of DOM from CPOM and pathways of FPOM generation are also shown. (Shredders--amphipod, detrital stonefly, caddisfly, and cranefly; filtering collectors--blackflies and net spinning caddisfly; gathering collectors--burrowing mayfly; scrapers--tortise-shell case caddisfly, limpet, heptageniid mayfly, waterpenny beetle larva; piercers--microcaddisflies; predators--predaceous stonefly, sulpin.)

is correlated with in-stream algal primary production, for example, P/R ratio (Anderson and Cummins, 1979). Ratios of shredders or collectors to scrapers are indicative of the importance of CPOM or FPOM relative to periphyton as nutritional resources.

The piercer:macrophyte association (Fig. 1) in streams is represented almost exclusively by microcaddisflies, which utilize filamentous macroalgae by sucking the fluids from individual cells. As primary producer communities in streams shift from diatoms to macrophytes, the ratio of piercers to scrapers increases. The piercers are a unique group in that the major utilization of macrophytes in streams is in detrital food chains (Minshall, 1978).

Predator:prey associations (Fig. 1) in stream communities appear to be relatively constant and ubiquitous. Animal tissue represents the highest quality food resource (Anderson and Cummins, 1979), but the relatively low density of prey relative to other nutritional resources means that predators are required to expend more energy in acquiring food.

MANAGEMENT CONSIDERATIONS

The multiple and intimate relationships between the riparian zone and the stream ecosystem in forested watersheds make this a critical interface for management. The riparian zone should be maintained as a suitable source area for long-term physical channel structure (e.g., wood debris) and annual organic resources. Tools are available for evaluating the stream community response to changes in the riparian source area, such as: the C/N of nutritional resources, community metabolism (P/R, and macroinvertebrate functional group ratios.

Because the quality and quantity of inputs to forested headwater stream ecosystems from the riparian zone exerts a major control on community structure and function, a number of management strategies are possible. For example, selective harvest or enhancement of tree, shrub, or herbaceous species in the riparian zone would be possible. Species such as alder generate rapidly processed litter which produces nitrogen-rich leachate that is quickly converted to FPOM, while conifer needles (e.g., Douglas-fir) are utilized at much slower rates over longer time periods.

In general, management of riparian zones is management of headwater streams, and management of headwater streams is critical for managing the larger receiving streams and rivers.

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Mass Soil Movement

Fred Swanson

I. Soil Creep

- A. Significance
- B. Method of monitoring
- C. Observed movement
- D. Management impact

II. Earthflows

- A. Significance
- B. Methods of monitoring
- C. Observed movement
- D. Management impact

III. Debris avalanches

- A. Significance
- B. Site specific engineering studies
- C. Studies of populations of events - <u>inventories</u>
  1. The value of slide inventories in forest management
  2. Developing an inventory
  3. An example from the Willamette National Forest
  4. Use of inventories - planning, monitoring

IV. Forest-Research cooperation

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10

TIMBER HARVESTING, MASS EROSION, AND STEEPLAND FOREST GEOMORPHOLOGY IN THE PACIFIC NORTHWEST

Douglas N. Swanston and Frederick J. Swanson

INTRODUCTION

Forest operations in mountainous regions of the Pacific Northwest have a major impact on soil-erosion processes. The mountains of the region are youthful, the area has undergone recent tectonic activity, and west of the crest of the Cascades annual precipitation may exceed 375 cm. Consequently, natural erosion rates are high. Heavy forest vegetation and the high infiltration capacity of many of the forest soils protect slopes from surface erosion. The combination of these factors results in mass erosion processes generally being the dominant mechanisms of sediment transport from hillslopes to stream channels. The principal mass erosion processes are slow, downslope movement involving subtle deformation of the soil mantle (creep) and discrete failures, including slow-moving, deep-seated slump—earthflows; rapid, shallow soil and organic debris movement from hillslopes (debris avalanches); and rapid debris movement along downstream channels (debris torrents) (Fig. 1A, B). In many areas, forest vegetation plays an important role in stabilizing slopes and reducing the movement rate and occurrence of these mass erosion processes. When timber is removed from marginally stable slopes, whether by natural processes such as wildfire and wind or by the activities of man, a temporary acceleration of erosion activity is likely.

Accelerated erosion due to forest land management activities may result in reduced productivity of forest soils over sizable portions of affected watersheds, damage to roads, bridges, and other structures, and adverse impacts on the stream environment downstream.

FACTORS CONTROLING MASS EROSION PROCESSES

Geologic, hydrologic, and vegetative factors control the occurrence and relative importance of mass erosion processes. In the Pacific Northwest, areas of clay-rich bedrock and deep, cohesive soils are characterized by dominance of the slow mass movement processes of creep and slump—earthflow; notably in the extensive areas of soft sedimentary rocks of the Klamath Mountains, Oregon, and northern California Coast Range and the volcaniclastic rocks in the Cascade Range. Debris avalanches dominate mass erosion processes in terrain typified by steep slopes, cohesionless soils, and relatively competent bedrock, such as large areas of the coast ranges of Oregon, Washington, British Columbia, and Alaska.

A

FIGURE 1 *Examples of principal mass erosion processes operating on steep forested slopes in the Pacific Northwest. (A)* creep *and* slump–earthflow *terrain on Franciscan sediments in the upper headwaters of the North Fork, Eel River, northern California. The entire slope is undergoing creep deformation but note the discrete failure (slump–earthflow) marked by the steep headwall scarp at top center and the many small slumps and debris avalanches triggered by surface springs and road construction. (B) debris avalanches developed in shallow soils overlying compact till in southeast Alaska.* Debris torrent *developed below debris avalanche at center of photo due to channeling of debris, undercutting of sideslopes, and addition of material by secondary avalanching into channel.*

Periods of high-intensity precipitation, storm events, and condensation snowmelt commonly trigger or accelerate mass wasting events on steep forest slopes (Bishop and Stevens, 1964; Fredriksen, 1965; Dyrness, 1967; Swanston, 1969). These factors directly influence the moisture content of the soil and determine the presence or absence of active piezometric levels in the subsurface. Moisture content and piezometric level affect the weight of the soil mass and control the development of positive pore pressures. These factors act to reduce the resistance of the soil mass to sliding by either mobilization of clay structures, primarily through adsorption of water into the clay mineral structure, or by reducing the frictional resistance of the soil mass along the failure surface.

Vegetation cover in general helps control the amount of water reaching the soil and the amount held as stored water, largely through a combination of interception and evapotranspiration. In high-rainfall areas of the Pacific Northwest, interception is negligible during large storm events important to mass soil movement (Rothacher, 1963). Evapotranspiration (ET) has its principal effect on soil strength by reducing the length of time that soils remain saturated. ET reduces soil moisture during dry months, reduces the degree of saturation that can result from the first storms of the fall recharge period, and accelerates the rate of soil moisture removal at the end of the wet season. Once the soil is recharged as the wet season begins, the effect of ET loss becomes negligible. Also of importance is the depth of withdrawals by ET. Deep withdrawals may require substantial recharge to satisfy the soil-water deficit, delaying the attainment of saturated soil conditions several months or through a number of major slide-producing events. Shallow soils, on the other hand, will recharge rapidly, possibly attaining saturated conditions and maximum instability during the first major storm. ET has also been linked with at least a temporary increase of shear strength and stability during dry summer months through soil moisture depletion and buildup of soil moisture stress (Gray and Brenner, 1970; Manbeian, 1973; Gray, 1973).

Forests may also moderate the rate at which moisture enters the soil. This is particularly important in the case of warm-rain-on-snow events in which forest vegetation may influence the amount of snow collected on the land surface and the rate of melting, where advection and condensation melting are important (Anderson, 1969).

A crucial factor in the stability of active slopes is the role of plant roots in maintaining the shear strength of soil mantles. Roots add strength to the soil by vertical anchoring through the soil mass into fractures in the bedrock, and by laterally tying the slope together across zones of weakness or instability. In shallow soils, both effects may be important. In deep soils the vertical rooting factor will become negligible, but lateral anchoring may remain important. In some steep areas in the western United States, rooting strength may be the dominant factor in maintaining the slope equilibrium of an otherwise unstable area (Bishop and Stevens, 1964; Croft and Adams, 1950). Zaruba and Mencl (1969) have reported the stabilizing effect of tree roots in landslide areas in Czechoslovakia, and Nakano (1971) reports similar effects from unstable areas in Japan.

IMPACTS OF FOREST-MANAGEMENT ACTIVITIES ON MASS EROSION PROCESSES

Timber-harvesting activities, including clearcutting and road construction, modify factors influencing mass erosion in a variety of ways. Some of the most important impacts of these activities are summarized in Table 1.

TABLE 1 *Impacts of Engineering Activities on Factors That Influence Slope Stability in Steep Forest Lands of the Pacific Northwest*

Factors	Engineering activities[a]: Deforestation	Engineering activities[a]: Roading	References
I. Hydrologic influences			
A. Water movement by vegetation	Reduce evapotranspiration (−)	Eliminate evapotranspiration (−)	Gray (1970) Brown and Sheu (1975)
B. Surface and subsurface water movement	Alter snowmelt hydrology (− or +) Alter concentrations of unstable debris in channels (−) Reduce infiltration by ground surface disturbance (−)	Alter snowmelt hydrology (− or +) Alter surface drainage network (−) Intercept subsurface water at roadcuts (−) Alter concentrations of unstable debris in channels (−) Reduce infiltration by roadbed (−)	Anderson (1969) Harr et al. (1975) Megahan (1972) Rothacher (1959) Froehlich (1973)
II. Physical influences			
A. Vegetation			Swanston (1970)
1. Roots	Reduce rooting strength (−)	Eliminate rooting strength (−)	Nakano (1971)
2. Bole and crown	Reduce medium for transfer of wind stress to soil mantle (+)	Eliminate medium for transfer of wind stress to soil mantle (+)	Swanston (1969)
B. Slope			
1. Slope angle		Increase slope angle at cut and fill slopes (−)	Parizek (1971), O'Loughlin (1972)
2. Mass on slope	Reduce mass of vegetation on slope (+)	Eliminate mass of vegetation on slope (+) Cut and fill construction redistributes mass of soil and rock on slope (− or +)	Bishop and Stevens (1964) O'Loughlin (1972)
C. Soil properties		Reduce compaction and apparent cohesion of soil used as road fill (−)	

[a]Influence that usually increases slope stability denoted by (+); influence that usually decreases stability denoted by (−).

The principal impacts of forest removal by clearcutting are to reduce rooting strength and to alter the hydrologic regime at the site. In Japan, Kitamura and Namba (1966, 1968) have described a period of greatly reduced soil strength attributable to rooting beginning about 3 yr following cutting, when there has been significant decay of root systems, and attaining a minimum strength 15 yr after cutting. Similar loss of roting strength on unstable sites has been reported from coastal Alaska[1] and British Columbia (O'Loughlin, 1974), with minimum rooting strength attained 3 to 5 yr after cutting.

The hydrologic impacts of clearcutting include modification of annual soil-water status and changes in peaks of soil water held in detention storage during periods of storm runoff. Increased peak flows can generate active pore-water pressures, triggering shallow debris avalanches and debris torrents. Reduced ET due to clearcutting results in the soil-water status remaining at higher levels for several months longer than it would under forested conditions (Gray, 1970; Rothacher, 1971). This modification in the soil-water regime may result in prolonged periods of active creep and slump—earthflow movement during a single season or reactivation of dormant terrain. Water-yield studies in experimental watersheds in Oregon (Rothacher, 1971; Harr, 1975) suggest that this effect may continue for more than a decade after cutting.

Timber removal may also increase peaks of soil water by accelerated snowmelt during warm-rain-on-snow conditions (Anderson, 1969). This phenomenon may also increase total surface runoff if rain and snowmelt are synchronized (Rothacher and Glazebrook, 1968).

The principal impacts of road construction are to interrupt the natural balance between the resistance of the soil to failure and the downslope stress of gravity by disturbance of marginally stable slopes and alteration of subsurface and surface-water movement. Disturbance results from careless or improper cutting of marginally stable slopes, poor construction and placement of fills on steep slopes, and improper drainage design. Roads alter the routing of water by interception of surface water at cut slopes and surface drainage from roads and by carrying this excess water through ditches (Megahan, 1972; Harr et al., 1975). Mass erosion commonly occurs where natural and artificial drainage systems are inadequate to handle this excess water.

CREEP

Characteristics

Creep is the slow, downslope movement of the soil mantle in response to gravitational stress. The mechanics of creep have been investigated experimentally and theoretically by a number of workers (Terzaghi, 1953; Goldstein and Ter-Stepanian, 1957; Saito and Uezawa, 1961; Culling, 1963; Haefeli, 1965; Bjerrum, 1967; Carson and Kirkby, 1972; and others). Movement is by quasi-viscous flow, occurring under shear stresses sufficient to produce permanent deformation, but too small to result in discrete failure. Mobilization of the soil mass is primarily by deformation at grain boundaries and within clay mineral structures. Both interstitial and absorbed water appear to contribute to creep movement by opening the structure within and between mineral grains, thereby reducing

[1] Douglas N. Swanston and W. J. Walkotten, tree rooting and soil stability in coastal forests of southeastern Alaska. Study No. FS-NOR-1604:26 on file at PNW Forestry Sciences Laboratory, Juneau, Alaska.

friction within the soil mass. This permits a "remolding" of the clay fraction, transforming it into a slurry, which then lubricates the remaining soil mass. In local areas where shear stresses are great enough, discrete failure may occur, resulting in development of slump–earthflow due to progressive failure of the mantle materials.

Movement Rate and Occurrence

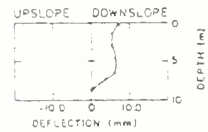
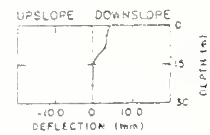
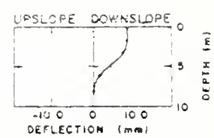
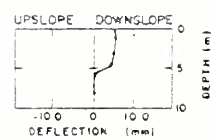
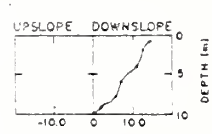
Creep movement generally occurs at rates of a few millimeters to a few centimeters per year. Therefore, long periods of observation and moderately sophisticated instruments are necessary to characterize creep. For these reasons, creep research has been focused on applied engineering problems in major construction activities (e.g., Wilson, 1970), and there have been few studies in the forest environment (Kojan, 1968; Barr and Swanston, 1970; Swanston, unpublished data). Recent measurements of creep rate using an inclinometer in flexible plastic pipe are summarized in Table 2. Also shown are examples of creep velocity profiles through soil masses.

The data shown in Table 2 are preliminary values based on 2 to 3 yr of monitoring and may not adequately represent long-term creep rates for these sites. Significant creep has been observed, however, and some interesting indications of the character of natural creep activity are apparent. Natural creep rates monitored in different geological materials in the western Cascade Range and coast ranges of Oregon and northern California indicate rates of movement between 7.1 and 15.2 mm/yr. The zone of most rapid movement usually occurs at or near the surface, although a zone of maximum displacement is usually present at depths associated either with incipient failure planes or zones of groundwater movement. The depth over which creep is active is quite variable and is largely dependent on parent material origin, degree and depth of weathering, subsurface structure, and soil-water content.

Movement rates are variable, as would be expected under natural conditions, but as a general rule lie within the range of 0 to 15 mm/yr. The maximum measured creep rates in Table 2 average 10.1 mm/yr. At many of the sites, movement takes place primarily during the rainy season when maximum soil-water levels occur (Fig. 2A), although creep may remain constant throughout the year in areas where the water table does not undergo significant seasonal fluctuation (Fig. 2B). This is consistent with Ter-Stepanian's (1963) theoretical analysis, which showed that the downslope creep rate of an inclined soil layer was exponentially related to the piezometric level in the slope.

Creep is generally the most persistent of all mass erosion processes. It operates at varying rates in clayey soils at slope angles of even just a few degrees. Therefore, in small watersheds developed in cohesive materials, creep may be operating over more than 90% of the landscape. The result of this creep activity is a continuing supply of soil material to the stream in the form of encroaching banks and small-scale bank failures. The quantity of soil delivered is quite large, and the supply is continuous from year to year. For example, assuming a creep rate of 10 mm/yr moving mantle material with a dry unit weight of 1,600 kg/m^3 to a stream with a bank approximately 2 m high (conservative estimates for watersheds in pyroclastic materials within the western Cascades Range, Oregon), approximately 64 metric tons/lineal km/yr will be supplied to the channel annually. During high-flow events, this material is carried into the stream by direct water erosion and by undercutting and local bank slumping. Such processes have been demonstrated to be a major contributor to sediment loads of the Eel and Mad rivers in northern

TABLE 2 *Examples of Measured Rates of Natural Creep on Forested Slopes in the Pacific Northwest*

				Maximum downslope creep rate		
Location	Data source	Parent material	Depth of significant movement (m)	Surface (mm/yr)	Zone of accelerated movement (mm/yr)	Representative creep profile
Coyote Creek South Umpqua River drainage, Cascade Range of Oregon, site C-1	Swanston[a]	Little Butte Volcanic Series: deeply weathered clay-rich andesitic dacitic volcaniclastic rocks	7.3	13.97	10.9	
Blue River drainage—Lookout Creek H. J. Andrews Exp. Forest, Central Cascades of Oregon, site A-1	Swanston[a]	Little Butte Series: (same as above)	5.6	7.9	7.1	
Blue River drainage IBP Experimental Watershed 10, site No. 4	McCorison[b] and Glenn	Little Butte Volcanic Series	0.5	9.0	—	
Baker Creek, Coquille River, Coast Range, Ore., Site B-3	Swanston[a]	Otter Point Formation: highly sheared and altered clay-rich argillite and mudstone	7.3	10.4	10.7	
Bear Creek, Nestucca River, Coast Range, Ore., site N-1	Swanston[a]	Nestucca Formation: deeply weathered pyroclastic rocks and interbedded, shaly siltstones and claystones	15.2	14.9	11.7	
Redwood Creek, Coast Range, northern Calif., site 3-B	Swanston[a]	Kerr Ranch Schist: sheared, deeply weathered clayey schist	2.6	15.2	10.4	

[a]Douglas N. Swanston, unpublished data on file at Forestry Sciences Laboratory, U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station, Corvallis, Ore.
[b]F. Michael McCorison and J. F. Glenn, data on file at Forestry Sciences Laboratory, U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station, Corvallis, Ore.

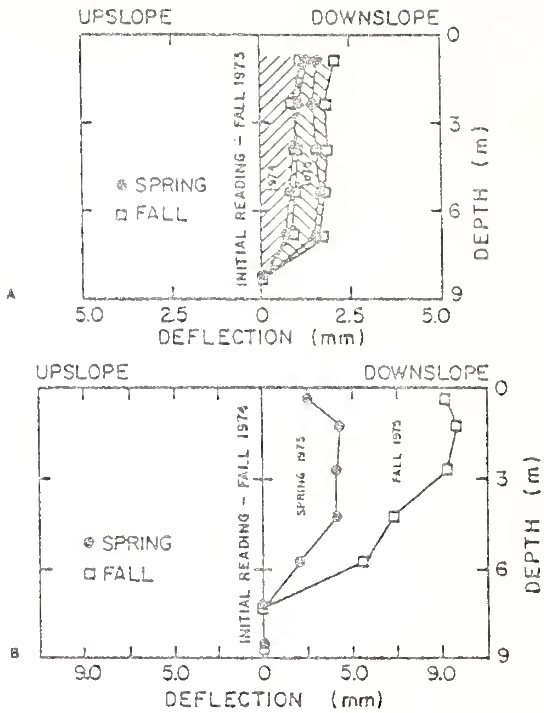


FIGURE 2 *Deformation of inclinometer tubes at two sites in the southern Cascade Range and Coast Range of Oregon. (A) Coyote Creek in the southern Cascade Range showing seasonal variation in movement rate as the result of changing soil-water levels. Note that the difference in readings between spring and fall of each year (dry months) is very small. (B) Baker Creek, Coquille River, Oregon Coast Range, showing constant rate of creep as a result. of continual high-water levels.*

California, which have the largest sediment discharges in the world. Anderson (1971) cites estimates that 80 to 85% of the total volume of sediment produced by these two rivers is the result of landslide and streambank erosion. In areas characterized by low-flow conditions supplemented only occasionally by storm flows, creep may fill the channel with soil and debris, and the stream water may be carried by subsurface flow and piping within the channel filling. Only during storm periods is flow great enough to open the channel and remove the debris stored there, resulting in the periodic discharge of excessive sediment loads from affected streams, and occasional torrent flow occurrence where local damming by debris occurs. Such a mechanism dominates in the great majority of smaller watersheds in the creep-dominated areas of the Pacific Northwest.

Impact of Forest Operations

There have been no direct measurements of the impact of harvesting activities on creep rates in the forest environment, mainly because of the long periods of record needed both before and after a disturbance. However, there are a number of indications that creep rates are accelerated by clearcutting and road construction.

Wilson (1970) and others have used inclinometers to verify accelerated creep following modification of slope angle, compaction of fill materials, and redistribution of soil mass at construction sites. The common occurrence of shallow-soil mass movements in these disturbed areas and open tension cracks along roadways at cut and fill slopes suggest that similar features along forest roads are indicators of significantly accelerated creep movement.

On slopes where clearcutting is the principal influence, impact on creep rates may be more subtle, involving modifications of hydrology and root strength. Where creep is a

shallow phenomenon (less than 2 to 3 m), the loss of root strength due to clearcutting is likely to be significant. Reduced evapotranspiration following clearcutting (Gray, 1970; Rothacher, 1971) may result in greater duration of the annual period of creep activity and, thereby, increase the annual creep rate.

Brown and Sheu (1975) have developed a mathematical model of creep that accounts for root strength, wind stress on the soil mantle, weight of vegetation, and soil moisture. The model predicts a brief period of increased slope stability following clearcutting due to removal of the weight of vegetation and elimination of wind stress. Thereafter, creep rates are accelerated as soil strength is attenuated by progressive root decay. The net result of the short-term increase and longer-term decrease in slope stability is expected to be an overall increase in creep activity.

SLUMP–EARTHFLOW

Characteristics

In local areas where shear stresses are great enough, discrete failure occurs and slump–earthflow features (Varnes, 1958) are formed. Simple slumping takes place as a rotational movement of a block of earth over a broadly concave slip surface and involves very little breakup of the moving material. Where the moving material slips downslope and is broken up and transported either by a flowage mechanism or by gliding displacement of a series of blocks, the movement is termed slow earthflow (Varnes, 1958). The combined term slump–earthflow is used because many deep-seated mass movements in the Pacific Northwest have slump characteristics in the headwall area and develop earthflow features downslope.

Slump–earthflows have been described by Varnes (1958), Wilson (1970), Colman (1973), Swanson and James (1975), and others. In the Pacific Northwest, these features may range in area from less than 1 hectare to more than several square kilometers. The zone of failure occurs at depths of a few meters up to several tens of meters below the surface. Commonly, there is a slump basin with a headwall scarp at the top of the failure area. Lower ends of earthflows typically run into stream channels. Transfer of earthflow debris to stream channels may take place by shallow, small-scale debris avalanching or by gullying and surface erosion, depending on soil and vegetation conditions. Therefore, the general instability set up by an active slump–earthflow initiates erosion activity by a variety of other processes.

Geologic, vegetative, and hydrologic factors have primary control over slump–earthflow occurrence. Deep, cohesive soils and clay-rich bedrock are especially prone to slump–earthflow failure, particularly where these materials are overlain by hard, competent rock (Wilson, 1970; Swanson and James, 1975). Earthflow movement also appears to be most sensitive to long-term fluctuations in the amount of available soil water (weeks, months, annually) (Wilson, 1970; and others).

Because earthflows are slow moving, deep-seated, poorly drained features, individual storm events probably have much less influence on their movement than on the occurrence of debris avalanches and torrents. Where planes of slump–earthflow failure are more than several meters deep, weight of vegetation and vertical root-anchoring effects are negligible (O'Loughlin, 1974).

Movement Rate and Occurrence

Movement rates of earthflows vary from imperceptibly slow to more than 1 m/day in extreme cases. In parts of the Pacific Northwest, many slump–earthflow areas appear to be presently inactive (Colman, 1973; Swanson and James, 1975). Areas of active movement may be recognized by fresh ground breaks at shear and tension cracks and by tipped and bowed trees. Rates of movement may be monitored directly by repeated surveying of marked points, by inclinometers, and by measuring deflection of roadways and other reference systems. These methods have been used to estimate the rates of earthflow movement shown in Table 3. However, these average rates, stated so simply, are somewhat misleading, because of the great variability of movement rate over both time and space, even for a single slump–earthflow (Colman, 1973; Swanson and James, 1975). Open tension cracks and degree of disturbance of vegetation on slump–earthflows indicate that some part of an earthflow terrane may move rather rapidly, while other areas appear to be temporarily stabilized.

The history of individual slump–earthflows may extend over thousands of years. This is indicated by age estimates based on radiometric dating of included wood, calculation of volume of eroded material, estimated long-term rate of earthflow erosion from site, presence of 7,000-yr-old Mazama ash on preexisting earthflow terrain, and characteristics of drainage development over earthflow surfaces. During this long history, periods of relatively high precipitation or forest removal may increase water content of an earthflow and accelerate movement rate. This downslope movement will decrease the relief at the site until the stability of the mass is increased to a point where the velocity decreases markedly. A period of temporary inactivity will take place until there is reactiviation during a period of high moisture availability, or when the area had been destabilized by stream erosion at the toe of the earthflow.

The areal occurrence of slump–earthflows is mainly determined by bedrock geology. In the Redwood Creek Basin, northern California, Colman (1973) observed that, of the 27.4% of the drainage which is in slumps, earthflows, and older or questionable landslides, a very high percentage of the unstable areas are located in the clay-rich and

TABLE 3 *Observations of Movement Rates of Four Active Earthflows in the Western Cascade Range, Oregon*

Location	Period of record (yr)	Movement rate (cm/yr)	Method of observation
Landes Creek (Sec. 21 T22S R4E)	15	12	Deflection of road
Boone Creek (Sec. 17 T17S R5E)	2	25	Deflection of road
S.W. Cougar Reservoir (Sec. 29 T17S R5E)	2	2.5	Deflection of road
Lookout Creek (Sec. 30 T15S R6E)	1	7	Strain rhombus measurements across active ground breaks

pervasively sheared portions of the Franciscan assemblage of rocks. Areas underlain by schists and other more highly metamorphosed rock are much less prone to deep-seated mass erosion. The areal occurrence of slump–earthflows in volcanic terranes of the Pacific Northwest is also closely linked to bedrock type. In the H. J. Andrews Experimental Forest, western Cascade Range, Oregon, for example, approximately 25.6% of areas underlain by volcaniclastic rocks are included in active and presently inactive slump–earthflows. Less than 1% of areas of basalt and andesite flow rock have undergone slump–earthflow failure.

Impact of Forest Operations

Engineering activities that involve excavation and fills frequently have dramatic impact on slump–earthflow activity (Wilson, 1970). In the forest environment, there are numerous unpublished examples of accelerated or reactivated slump–earthflow movement after forest road construction. Undercutting of toe slopes of earthflows and piling of rock and soil debris on slump blocks are common practices that increase slump–earthflow movement. Stability of such areas is also affected by modification of drainage systems, particularly where road drainage systems route additional water into the slump–earthflow areas. These disturbances may increase movement rates from a few millimeters per year to several tens of centimeters per year or more. Once such areas have been destabilized, they may continue to move at accelerated rates for several years.

Although the impact of clearcutting alone on slump–earthflow movement has not been demonstrated quantitatively, several pieces of evidence suggest that it may be significant. In massive, deep-seated failures, lateral and vertical anchoring of tree-root systems is negligible. However, hydrologic impacts appear to be important. Increased moisture availability due to reduced evapotranspiration will increase the volume of water not utilized by the vegetation. This water is therefore free to pass through the rooting zone to deeper levels of the earthflow. Although the hydrology of slump–earthflows has not yet been investigated, hydrology research on small watersheds suggests that this effect may be substantial. For example, even 9 yr after clearcutting of a small watershed in the H. J. Andrews Experimental Forest, runoff may still exceed by 50 cm the estimated yield for the watershed in a forested condition (Rothacher, 1971; Harr, 1975, pers. comm.). In watersheds with steep slopes and relatively permeable soils, this increased water yield comes as higher base flow during dry months and higher peak and base flow during early and late stages of the wet season (Rothacher, 1971). On poorly drained earthflows, the increased available moisture is likely to be stored in the subsoil for longer periods of time, possibly contributing to increased rate and duration of the wet season earthflow movement. It is not known whether possible clearcutting-related increases in peak discharge of surface and subsurface water influences earthflow movement.

DEBRIS AVALANCHES

Characteristics

Debris avalanches are rapid, shallow-soil mass movements from hillslope areas. Here we use the term "debris avalanche" in a general sense encompassing debris slides, avalanches, and flows, which have been distinguished by Varnes (1958) and others on the basis of increasing water content. From a land-management standpoint, there is little

purpose in differentiating failures among these types of shallow hillslope, since the mechanics and controlling and contributing factors are the same, and one frequently leads to another.

Debris avalanches have rather consistent characteristics in the variety of geologic and geomorphic settings extending from northern California to southeast Alaska (Colman, 1973; Morrison, 1975; Dyrness, 1967; Gonsior and Gardner, 1971; Fiksdal, 1974; O'Loughlin, 1972; Bishop and Stevens, 1964; Swanston, 1970). In all these areas, debris avalanches are usually triggered by infrequent, intense storms. For example, in the H. J. Andrews Experimental Forest, Oregon, it has taken storms of a 7-yr return period or greater to initiate debris avalanching in forested areas. Swanston (1969) has correlated storms with a 5-yr return interval with accelerated debris avalanching in coastal Alaska.

Debris avalanches leave scars in the form of spoon-shaped depressions from which less than 10 to more than 10,000 m^3 of soil and organic debris have moved downslope. Average volumes of individual debris avalanches in forested areas in the Pacific Northwest range from about 1,540 to 4,600 m^3.

Several of the factors discussed previously control the occurrence of debris avalanches. Debris-avalanche-prone areas are typified by shallow, noncohesive soils on steep slopes where subsurface water may be concentrated by subtle topography on bedrock or glacial till surfaces. Because debris avalanches are shallow failures, factors such as root strength, anchoring effects, and the transfer of wind stress to the soil mantle, are potentially important influences. Factors that influence antecedent soil moisture conditions and the rate of water supply to the soil by snowmelt and rainfall also have significant control over when and where debris avalanches occur.

Movement Rate and Occurrence

Movement rates of debris avalanches have seldom been measured because of the extreme storm conditions under which they occur. However, based on the few available accounts and the steep slope conditions where debris avalanches occur, their rates of movement probably range as high as 20 m/s.

The rate of occurrence of debris avalanches is controlled by the stability of the landscape and the frequency of storm events severe enough to trigger them. Therefore, the rates of erosion by debris avalanching will vary from one geomorphic–climatic setting to another. Table 4 shows that annual rates of debris-avalanche erosion from forested areas of study sites in Oregon, Washington, and British Columbia range from 11 to 72 $m^3/km^2/yr$. These estimates are based on surveys and measurements of erosion by each debris avalanche occurring in a particular time period (25 yr or longer) over a large area (12 km^2 or larger).

Impact of Forest Operations

The net impact of engineering activities can be estimated by determining the rate of debris-avalanche erosion in clearcut areas and road rights-of-way and comparing these levels of erosion with the erosion rate for forested areas during the same time period. Such an analysis (Table 4) reveals that clearcutting commonly results in acceleration by a factor of 2 to 4 of debris-avalanche erosion. Roads appear to have a much more profound impact on erosion activity. In the four study areas listed in Table 4, road-related debris-avalanche erosion was increased by factors ranging from 25 to 340 times the rate of debris-avalanche erosion in forested areas.

TABLE 4 *Debris-Avalanche Erosion in Forest, Clearcut, and Roaded Areas*

Site	Period of record (yr)	Area (%)	Area (km^2)	Number slides	Debris-aval. erosion ($m^3/km^2/yr$)	Rate of debris-avalanche erosion relative to forested areas
Stequaleho Creek, Olympic Peninsula (Fiksdal, 1974)						
Forest	84	79	19.3	25	71.8	× 1.0
Clearcut	6	18	4.4	.0	0	0
Road R/W	6	3	0.7	83	11,825	×165
			24.4	108		
Alder Creek, western Cascade Range, Oregon (Morrison, 1975)						
Forest	25	70.5	12.3	7	45.3	× 1.0
Clearcut	15	26.0	4.5	18	117.1	× 2.6
Road R/W	15	3.5	0.6	75	15,565	×344
			17.4	100		
Selected Drainages, Coast Mountains, S.W. British Columbia (O'Loughlin, 1972, and pers. comm.)						
Forest	32	88.9	246.1	29	11.2	× 1.0
Clearcut	32	9.5	26.4	18	24.5	× 2.2
Road R/W	32	1.5	4.2	11	282.5[a]	× 25.2
			276.7	58		
H. J. Andrews Experimental Forest, western Cascade Range, Oregon (Swanson and Dyrness, 1975)						
Forest	25	77.5	49.8	31	35.9	× 1.0
Clearcut	25	19.3	12.4	30	132.2	× 3.7
Road R/W	25	3.2	2.0	69	1,772	× 49
			64.2	130		

[a]Calculated from O'Loughlin (1972, and pers. comm.), assuming that area involving road construction in and outside clearcuts is 16% of area clearcut.

The great variability of the impact of roads reflects not only differences in the natural stability of the landscapes, but also differences in road-location design and construction. For example, the Alder Creek and H. J. Andrews Experimental Forest study sites are in similar climatic, geologic, and geomorphic settings, both areas are managed by a single National Forest, and the level of debris-avalanche erosion in forested areas of the two drainage is similar. However, the level of road-related debris-avalanche erosion in the Alder Creek drainage has been nearly eight times greater than in the Andrews Forest. This contrast in road impact appears to result from higher proportions of midslope road mileage and large, unstable hillslopes in the Alder Creek area, which was managed in a less stringent fashion than the experimental forest (Morrison, 1975).

The duration of impacts of clearcutting and road construction on debris-avalanche erosion is not well documented. Ideally, it would be useful to know how the various factors influencing slope stability, such as rooting strength and soil-moisture storage, each vary as a function of time since disturbance. However, such information is known only in a qualitative sense.

The duration of the net effects of harvesting activities can be deduced from historical records such as those for the H. J. Andrews Experimental Forest, where the history of debris avalanches and associated clearcutting and road construction since 1950 has been documented (Swanson, unpublished data). Based on these observations, clearcut slopes appear to undergo a period of increased susceptibility to debris avalanching for about 12 yr after cutting. However, the strength of the conclusions that can be drawn from a record of only 25 yr is limited by the irregular, episodic nature of both storms and harvesting activities. The history of road-related debris avalanches with respect to road age is further complicated by the disproportionately high levels of natural instability in areas where road access has not yet been developed and by the changing standards of road construction and maintenance. These factors contribute to the very irregular patterns of road age at the time of failure shown in Table 5. It is important to note that roads in the two study areas (Table 5) continued to be very active sites of debris-avalanche erosion for more than 16 yr after initial construction of the road. Therefore, the duration of impact of road construction on erosion by debris avalanches may persist more than twice as long as clearcutting impacts.

The relative impacts of clearcutting and road construction on the total level of accelerated erosion are not clearly reflected in the data in Table 4. For example, in the H. J. Andrews Experimental Forest, roads accelerate debris-avalanche erosion to a much greater extent than clearcutting, but road rights-of-way cover much less area of the forest than do clearcut units. When road and clearcutting impacts are weighted by the area influenced by each activity, the two types of forest engineering activities contribute about equally to the total level of accelerated debris-avalanche erosion (Swanson and Dyrness, 1975).

DEBRIS TORRENTS

Characteristics

Debris torrents involve the rapid movement of water-charged soil, rock, and organic material down steep stream channels. Debris torrents are distinguished from debris

TABLE 5 *History of Debris Avalanches with Respect to Road Age*

Road age (yr)	Percent of debris avalanches: H. J. Andrews Exp. Forest[a]	Percent of debris avalanches: Clearwater Natl. Forest[b]
0–5	57	5
6–10	11	50
11–15	20	34
16+	12	11

[a]Swanson, unpublished data, based on a complete 25-yr history of road construction and 73 debris avalanches.

[b]Day and Megahan, 1975, based on debris avalanches occurring in a single large storm during Jan. 1974.

avalanches because the two types of mass movement events occur over different parts of the landscape; consequently, they have differing implications for the land manager.

Debris torrents typically occur in steep intermittent first- and second-order channels. These events are triggered during extreme discharge events by slides from adjacent hillslopes, which enter a channel and move directly downstream, or by the breakup and mobilization of debris accumulations in the channel. The initial slurry of water and associated debris commonly entrains large quantities of additional inorganic and living and dead organic material from the stream bed and banks. Some torrents are triggered by debris avalanches of less than 100 m^3, but ultimately involve 10,000 m^3 of debris entrained along the track of the torrent. As the torrent moves downstream, hundreds of meters of channel may be scoured to bedrock. When a torrent loses momentum, there is deposition of a tangled mass of large organic debris in a matrix of sediment and fine organic material covering areas up to several hectares.

The main factors controlling the occurrence of debris torrents are the quantity and stability of debris in channels, steepness of channel, stability of adjacent hillslopes, and peak discharge characteristics of the channel. The concentration and stability of debris in channels reflects the history of stream flushing and the health and stage of development of the surrounding timber stand (Froehlich, 1973). The stability of adjacent slopes is dependent on a number of factors described in previous sections on other mass erosion processes. The history of storm flows has a controlling influence over the stability of both soils on hillslopes and debris in stream channels.

Movement Rate and Occurrence

Although debris torrents pose very significant environmental hazards in mountainous areas of the Pacific Northwest, they have received little study (Fredriksen, 1963, 1965; Morrison, 1975; Swanson and Lienkaemper, 1975). Velocities of debris torrents, estimated to be up to several tens of meters per second are known only from verbal and a few written accounts. The occurrence of torrents has been systematically documented in only two small areas of the Pacific Northwest, both in the western Cascade Range of Oregon (Morrison, 1975; Swanson, unpublished data). In these studies, rates of occurrence of debris torrents were observed to be 0.005 and 0.008 events/km^2/yr for forested areas (Table 6). Torrent tracks initiated in forest areas ranged in length from 100 to 2,280 m and averaged 610 m of channel length. Debris avalanches have played a dominant role in triggering 83% of all inventoried torrents. Mobilization of stream debris not immediately related to debris avalanches has been a rather minor factor in initiating debris torrents in these Cascade Range streams. Therefore, the susceptibility of an area to debris avalanching is a direct indicator of the potential for debris torrents.

Impact of Forest Operations

Timber-harvesting activities appear to dramatically accelerate the occurrence of debris torrents by increasing the frequency of debris avalanches. Although it has not been demonstrated, it is also possible that increased concentrations of unstable debris in channels during harvesting (Rothacher, 1959; Froehlich, 1973; Swanson and Lienkaemper, 1975) and possible increased peak discharges (Rothacher, 1973; Harr et al., 1975) may accelerate the frequency of debris torrents.

The relative impacts of these factors and of clearcutting and roads may be assessed by using the frequency of occurrence of debris torrents (events/km^2/yr) in forest areas as an

estimate of the natural, background level of debris-torrent activity against which to compare the rates of occurrence attributed to roads and clearcutting (Table 6). In the H. J. Andrews Experimental Forest and the Alder Creek study sites, clearcutting appeared to increase the occurrence of debris torrents by 4.5 and 8.8 times; roads were responsible for increases of 42.5 and 133 times.

Although the quantitative reliability of these estimates of harvesting impacts is limited by the small number of events analyzed, there is clear evidence of marked increase in the frequency of debris torrents as a result of clearcutting and road building. The history of debris avalanches in the two study areas clearly indicates that increased debris torrents are primarily a result of two conditions: debris avalanches trigger most debris torrents (Table 6), and the occurrence of debris avalanches is greatly increased by clearcutting and road construction (Table 4).

The data in Table 6 suggest that clearcutting in the H. J. Andrews Experimental Forest may have had a significant impact on frequency of debris torrents. In several cases, increased concentrations of debris in streams appeared to have contributed to debris-torrent occurrence. The extent of management impact on stream debris concentrations is directly related to how carefully engineering activities are designed and carried out. Under different geomorphic conditions and management practices, logging-related increases of debris in streams may have a more dramatic impact on debris-torrent occurrence.

The close relationship between occurrence of debris avalanches and debris torrents also indicates that the duration and the relative impacts of roads and clearcutting on debris torrents and debris avalanches (discussed previously) will be very similar in timing and magnitude.

RELATIONSHIPS AMONG PROCESSES

Creep, slump–earthflows, debris avalanches, and debris torrents function as primary links in the natural transport of soil material to streams in the Pacific Northwest. The importance of the linkage among these processes and the apparent impact of timber harvesting on rate of movement and sediment yield to streams is illustrated by results from ongoing investigations at Coyote Creek in the South Umpqua Experimental Forest and in the H. J. Andrews Experimental Forest, both in Oregon.

Coyote Creek Erosion Research

The Coyote Creek research area is located approximately 65 km southeast of Roseburg, Oregon, in the western Cascade Range. Four research watersheds have been established on deeply weathered volcaniclastic materials of the Little Butte Formation. Since 1966, both streamflow and sediment discharge have been monitored in these watersheds to determine water yield and nutrient outflow under forested and clearcut conditions.[1] Two access roads were constructed across the upper half of watershed 3 in 1971, and the watershed was logged by clearcutting in 1972.

From 1966 to 1970, total bedload export, estimated from volumes of sediment removed from a weir at the mouth of the watershed, was much less than 0.01 m^3/m of

[1] Study 1602-10, A Study of the Effects of Timber Harvesting on Small Watersheds in the Sugarpine, Douglas-Fir Area of Southwestern Oregon, U.S. Department of Agriculture Forest Service, Forestry Sciences Laboratory, Corvallis, Ore.

TABLE 6 *Characteristics of Debris Torrents with Respect to Debris Avalanches and Landuse Status of Site of Initiation in the H. J. Andrews Experimental Forest (Swanson, Unpublished Data) and Alder Creek Drainage (Morrison, 1975)*

Site	Area of watershed (km^2)	Period of record (yr)	Debris torrents triggered by debris avalanches	Debris torrents with no associated debris avalanche	Total		Rate of debris torrent occurrence relative to forested areas
					No.	No./km^2/yr	
H. J. Andrews Experimental Forest, western Cascade Range, Oregon							
Forest	49.3	25	9	1	10	0.008	× 1.0
Clearcut	12.4	25	5	6	11	0.036	× 4.5
Road	2.0	25	17	—	17	0.340	× 42.0
Total	64.2		31	7	38		
Alder Creek drainage, western Cascade Range, Oregon							
Forest	12.3	90	5	1	6	0.005	× 1.0
Clearcut	4.5	15	2	1	3	0.044	× 8.8
Road	0.6	15	6	—	6	0.667	×133.4
Total	17.4		13	2	15		

TABLE 7 *Estimated Annual Bedload Export from Clearcut Coyote Creek Watersheds 3 and 4*

Year	Precipitation (cm)	Total bedload volume $(m)^3$	Volume/unit stream length (m^3/m)	Basin condition
1966	120.5	3.9	0.0091	Forested
1967	118.5	0.5	0.0012	Forested
1968	87.6	0.6	0.0014	Forested
1969	110.9	0.2	0.0005	Forested
1970	116.3	1.3	0.0030	Forested
1971	155.7	21.9	0.051	Roads
1972	153.3	69.9[a]	0.163	Clearcut
1973	89.5	3.2	0.0074	Clearcut
1974	156.5	46.4	0.108	Clearcut
1975	122.6	7.7	0.018	Clearcut

[a]Minimum estimate; basin overflowed.

active stream channel/yr $(m^3/m/yr)$ (Table 7; R. L. Fredriksen, pers. comm.). In 1971, sediment yield dramatically increased to approximately 0.05 $m^3/m/yr$ as the result of unusually heavy winter precipitation and increased runoff from the construction of logging access roads. The bedload materials were derived primarily from debris avalanching and rotational slumping along the banks of the stream draining the watershed. In 1972, the first year after the entire watershed was clearcut and also a year of exceptionally heavy rainfall, bedload movement tripled over prelogging and road-building levels to an estimated volume of 0.16 $m^3/m/yr$. During this period, two new debris avalanches reached the channel from midslope, possibly triggered by reduction of rooting strength of vegetation following the logging process. Part of this overall bedload increase is due to surface erosion of severely scarified soil resulting from piling of slash after logging. However, the greater part can be directly linked to mass movements along the channel.

The major increases in bedload deposition in the weir basins have occurred during storm periods. Reconnaissance of the area and dissection of the weir deposits immediately after a major storm during the winter of 1972 exposed layering of heterogeneous sedimentary materials separated by zones of organic accumulations, which defined short periods or pulses of heavy sediment deposition. Such pulses result from repeated episodes of slumping and debris avalanching into the channel above the weir. A survey of the channel above the weir showed that nine new bank slumps and debris avalanches had occurred as a result of the storm, two of which were large enough to provide the volumes of material necessary to fill the weir basin (approximately 5.4 m^3 each). After 1972, bedload yields have been much lower, but still substantially above prelogging levels. This reflects accelerated bank erosion and removal of stream-stored sediment due to increased peak flows. A detailed survey of the watershed has since revealed over 50 sites of active bank slumping and debris avalanching along the active stream channel, with volumes ranging from 5.6 to 350 m^3. Much of this material moved into the channel, diverting it or causing water to flow beneath the surface. At least 12 debris dams have blocked the channel, leading to temporary storage of from 45 to 1,340 m^3 of alluvium, soil, and organic debris behind each dam. In 1974, the total volume of material available for

stream transport was estimated to be 3,100 m^3. Of this, 1,090 m^3 was stored as slump blocks and 2,010 m^3 was stored behind debris dams.

In addition, quantitative creep measurements in the soil and deeply weathered volcaniclastic rocks in the lower half of the watershed indicate creep of the soil in the zone of maximum movement of approximately 10.9 mm/yr, with much of the movement taking place along a narrow zone of weakness at a depth of approximately 8.2 m (Table 2). Since the banks in the watershed average at least 1 m in height, soil material is supplied to the stream channel in the lower part of the watershed at annual rates of 0.02 m^3/m of active channel. There are approximately 430 m of active stream channel in watershed 3. Thus, approximately 8.6 m^3 of soil are made available for stream transport annually by creep movement. The average annual yield since 1972 has been 31.8 m^3/yr. This suggests that about 27% of the annual yield is being supplied by creep processes. The remainder is supplied by active slump–earthflows, surface erosion, and debris avalanching, or derived from stored channel deposits.

H. J. Andrews Experimental Forest

The H. J. Andrews Experimental Forest is located in the western Cascade Range of Oregon in an area characterized by lava flow and volcaniclastic bedrock (Swanson and James, 1975), average annual precipitation of 230 cm, and Douglas fir–western hemlock forest vegetation (Franklin and Dyrness, 1973). Erosion research in the forest has been focussed on two scales: (1) studies in small watersheds (less than 110 hectares) involving continuous monitoring of bedload and suspended sediment outflow since 1956 (Fredriksen, 1963, 1965, 1970), and (2) mapping and historical analysis of slump–earthflows, debris avalanches, and debris torrents throughout the forest (Dyrness, 1967; Swanson and James, 1975).

The history of mass erosion in the small watersheds and the entire experimental forest reveals the close interactions among the various mass erosion processes. As shown diagramatically in Figure 3, areas of creep and slump–earthflow activity may overlap, and these two processes contribute to the instability of areas that ultimately fail by debris avalanching. Debris avalanches, in turn, are dominant initiators of debris torrents.

Creep and slump–earthflow processes are interrelated in at least two senses. Creep deformation is thought to be a common precursor of slump–earthflows, occurring where strain in the form of creep has exceeded the shear strength of soil and rock material. Even where discrete failure has occurred and slump–earthflow movement has begun, superimposed creep deformation is likely to occur in the earthflow material. In one slump–earthflow terrain that is moving at the rate of about 5 cm/yr (Lookout Creek earthflow, Table 3), creep deformation within a single slump block of earthflow material has been measured at 7 mm/yr using an inclinometer tube.

Active slump–earthflow movement, which extends over only 3.3% of the entire forest, appears to have contributed to the instability leading to nearly 40% of the total volume of debris-avalanche erosion from forested areas in the past 25 yr. Slightly more than half of this earthflow generated erosion by debris avalanches occurred in streamside areas where earthflow movement constricted channels, and subsequent streambank cutting has led to rapid, shallow-soil mass movements. Much of the remaining debris-avalanche erosion in undisturbed areas has occurred in steep midslope areas where creep activity is an important destabilizing factor.

These close relationships among mass and fluvial erosion processes suggest that, if one

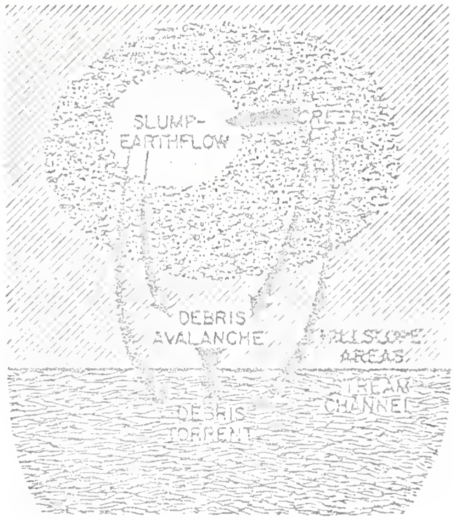


FIGURE 3 *Relationships among mass erosion processes. Arrows point from a process that sets up instability leading to failure by the process at head of the arrow. Width of arrows is a rough indication of degree of influence based on studies in the H. J. Andrews Experimental Forest.*

process is accelerated, the others may also be. Available data are insufficient to demonstrate the extent of impact of timber-harvesting activities on creep and slump–earthflow erosion in the H. J. Andrews Experimental Forest. However, in areas that have been clearcut and where roads have been built, debris-avalanche erosion has been increased by about five times (Swanson and Dyrness, 1975). This might be viewed as a reflection of accelerated creep and slump–earthflow activity in disturbed areas. Increased frequency of occurrence of debris avalanches also has resulted in a dramatic increase in debris torrents (Table 6).

CONCLUSIONS

Creep, slump–earthflows, debris avalanches, and debris torrents function as primary links in the natural transport of soil material to streams in the Pacific Northwest.

In areas characterized by deeply weathered, clay-rich mantle materials, creep movement may range as high as 15 mm/yr. Locally, where strain buildup causes discrete failure and the development of progressive slump–earthflows, transport rates of material to the stream may increase by several orders of magnitude. In areas characterized by steep slopes, shallow, coarse-grained mantle materials and steep, incised drainages, discrete failures producing debris avalanches and debris torrents transport large volumes of material to the stream at rates as high as 20 m/s.

Timber-harvesting operations, particularly clearcutting and road construction, accelerate these processes, the former by destroying the stabilizing influence of vegetation cover and altering the hydrologic regime of the site, the latter by interrupting the balanced strength–stress relationships existing under natural conditions by cut and fill activities, poor construction of fills, and alteration of surface and subsurface water movement.

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Erosion and Sediment Transport in Pacific Rim Steeplands. I.A.H.S. Publ. No. 132 (Christchurch, 1981)

Analysis of debris-avalanche erosion in steep forest lands:

An example from Mapleton, Oregon, USA

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Abstract

Inventories of shallow, rapid soil mass movements (debris avalanches) are useful for assessing impacts of forestry practices such as clearcutting and road construction. Analysis of inventory data from steep Pacific Rim forest land indicates a several-fold increase in debris-avalanche erosion from clearcutting relative to the rate in forested areas and even greater increases from roads. Some of the limitations of interpretation of inventory data can be overcome with improved data analysis.

Analyse de l'erosion par avalanche de debris en foret à pente raide: un exemple de Mapleton, Oregon, Etats-Unis.

Résumé

Les inventaires de mouvements superficiels rapides de terrain (avalanche de débris) sont utiles à l'étude des effets d'activités forestiares telles que le déboisement systématique et la construction de routes. L'analyse des données d'inventaires sur les terrains forestiers raides du rebord Pacifique indique un accroissement multiple de l'erosion par avalanche de débris due au déboisement systématique (par contraste avec les terrains boisés), et un accroissement encore plus grand dû aux routes. Certains problèmes dans l'interprétation des données d'inventaire peuvent être résolus par des améliorations dans l'analyse des données.

Shallow, rapid, soil mass movement is a common, if not predominant, erosion process in much of the steep mountain land of the Pacific Rim. These events, here termed debris avalanches, pose particular problems for forest-land managers who must consider their impacts on forest productivity, aquatic resources, human life, and structures. Rates of soil erosion by debris avalanches—and the effects of forest management practices on occurrence of these events—are commonly assessed with inventories of debris avalanches from forest, clearcuttings, and road rights-of-way. Results of such inventories may be used to (1) identify specific management activities in the past that have accelerated the occurrence of debris avalanches which can be reduced with improved practices, (2) assess the effectiveness of improved practices in reducing debris-avalanche erosion, and (3) predict impacts of future management activities. We discuss results and limitations of inventories as a means of assessing debris-avalanche erosion and related impacts of forest practices. Examples are mainly from the Mapleton Ranger District, Siuslaw National Forest, in the Coast Ranges of western Oregon, USA (Lat. 44 N, Long. 124 W).

The Mapleton area is characterized by steep, highly dissected, debris avalanche-prone slopes carved in gently dipping Tertiary graywacke sandstone and siltstone. Forests of 100- to 200-year-old Douglas-fir (Pseudotsuga menziesii (Mirb.) Franco) and western hemlock (Tsuga heterophylla (Raf.) Sarg.), younger stands of alder (Alnus rubra Bong.), and early successional vegetation in clearcuttings cover the area. About 40 per cent of the area has been clearcut in the past 30 years. Annual rainfall totals about 200 cm at the town of Mapleton and six-hour intensities in excess of 7.5 cm have been recorded. A three-day storm totalling 19.7 cm of rainfall in November 1975 triggered over

250 debris avalanches in the 83 400-ha District and led to a series of studies of debris-avalanche erosion (Swanson et al., 1977; Ketcheson, 1978; Gresswell et al., 1979).

Debris avalanches in the study area are typically initiated from sites at the head of incipient drainage depressions termed "hollows" by Dietrich and Dunne (1978). Slump and block-glide types of deeper, slower mass movement also occur and sometimes trigger debris avalanches.

METHODS

Estimation of the rate of debris-avalanche erosion is based on (1) area in which a set of events has occurred, (2) period in which they took place, and (3) volume of soil transported. For each land-use condition (forest, clearcut, road right-of-way), we estimated these three types of data for events that moved more than 7.6 m^3 of soil, using different combinations of field methods and interpretation of aerial photographs (color, scale = 1:15 840). Size distribution of debris avalanches were measured in the field because volumes of soil moved could not be accurately determined from aerial photographs. Analysis of photos, however, is an efficient way to inventory areal and temporal frequencies over large areas where time is determined by bracketing events between the dates of the photographs. These methods were used for clearcuttings and road rights-of-way. We analyzed events in forested areas using field traverses to locate debris-avalanche scars and dendrochronologic methods to date them. The steep, dissected terrain, heavy forest cover, and small size of debris-avalanche scars preclude effective use of aerial photographs for analysis of events in forests. Methods are discussed further by Swanson et al. (1977).

Our sampling concentrated on land type 47 (Soils Resource Inventory, Siuslaw National Forest), the most intensely dissected and steepest land in the Forest.

RESULTS AND DISCUSSION

Inventoried debris avalanches in forested areas of land type 47 occurred with a frequency of about 0.5 events km^{-2} yr^{-1}, average volume of 54 m^3, and a rate of soil transfer of 28 m^3 km^{-2} yr^{-1} (Table 1). The rate of soil transfer by debris avalanches in clearcuttings was 111 m^3 km^{-2} yr^{-1} or about 4 times the rate in forested areas. The rate for road rights-of-way was about 120 times greater than the forest rate. These observations are in general agreement with those of Ketcheson (1978) for land type 47 in the same area (Table 1). He measured a rate of debris-avalanche erosion of 11 m^3 km^{-2} yr^{-1} in forested areas and a rate 11 times higher in clearcuttings.

Comparison with the study of Gresswell et al. (1979) is difficult, because they examined the impact of a single storm rather than the history of debris avalanches over 10 or more years as did Swanson et al. (1977) and Ketcheson (1978). This difference in sampling period provides an opportunity to examine changes in effects of management activities with changing practices. Based on aerial photo interpretation of debris avalanches in clearcuttings for the period 1962-1972, Swanson et al. (1977) observed 0.34 events km^{-2} yr^{-1}, in contrast with 0.95 events km^{-2} in the 1975 storm (Gresswell et al., 1978). Road right-of-way events in 1957-1972 occurred at a rate of 4.12 km^{-2} yr^{-1}; the 1975 storm had 2.91 events km^{-2}. This apparent pattern of increasing occurrence in clearcuttings and a decrease in frequency of events from road rights-of-way may reflect the combined effects of (1) improved road siting, construction, and maintenance, particularly during storms when field crews attempt to maintain road drainage systems, and (2) increased proportion in successive years of clearcut area in the most unstable terrain, because easiest sites have been cut first.

Results of these studies in the Oregon Coast Ranges offer interesting comparisons with results of debris-avalanche inventories elsewhere in the Pacific Rim (Table 1). Rate of debris-avalanche erosion in clearcut areas at five of the six sites exceeds forest rates by 2.2 to 22 times. Rates from road rights-of-way exceed forest rates by 26 to 350 times. Most of the increase in soil transfer at disturbed sites is because of greater frequency rather than increased average volume of debris avalanches. Rates of debris-avalanche erosion in forested areas are about the same at all sites. Events in forest and clearcut areas of highly dissected terrain, such as the Oregon Coast Ranges and Notown, New Zealand (O'Loughlin and Pearce, 1976), are characterized by much higher frequency but lower average volume than the less-dissected areas. Apparently these intricately dissected areas provide many sites for potential failure, but thin soils and steep side slopes limit the volume of each event.

The high rate of debris-avalanche erosion from roads relative to clearcuttings is somewhat misleading, because roads affect much less of managed forest land than does clearcutting. If we assume that 6 per cent of an area is in road right-of-way, we can calculate the debris-avalanche erosion from roads as [(6 per cent of area X road erosion rate) - (6 per cent of area X forest erosion rate)]. This takes into account that some erosion may occur from the road area even if it is in forest. A similar calculation can be made for the 94 per cent of the area clearcut. These calculations for debris-avalanche data for all soil types in the Mapleton District indicate that clearcuttings contribute about 23 per cent of total accelerated debris-avalanche erosion in the area, and roads account for the remainder.

Several short- and long-term factors limit this means of estimating impacts of clearcuttings and roads on debris-avalanche erosion. The more important short-term (scale of decades) limitations center on (1) changes in management practices and type of landscape operated in through time, (2) temporal changes in proportion of land areas in forest, clearcuttings, and road rights-of-way, and (3) initial increase and subsequent decrease in rates of soil transfer in individual clearcut and road right-of-way areas. The last two factors cause estimates of management impact of the type shown in Table 1 to vary with length of record. The net effect is to underestimate initial impacts of management. These problems may be overcome with an accounting system that uses units of cumulative area per unit time (CAT units) such as hectare-years for clearcuttings and roads of different age classes. Rather than dividing total volume of soil transferred by (1) the area in each land-use class at the end of the record and (2) the period of record in years, the denominator should be CAT units, such as cumulative ha-yr in clearcuttings 0 to 5 years old. Such a system appropriately considers rates of soil transfer in clearcuttings and roads of different ages. The difficulty is that a full management history is required.

Limitations of these assessments of management impacts on debris-avalanche erosion over several rotations are (1) failure to account for the importance of major, natural, episodic forest disturbances, such as wildfire, in many ecosystems, and (2) the possibility that documented short-term increase in debris-avalanche erosion after clearcutting is simply because of changes in timing of erosion with no change in long-term erosion rate. Perhaps the 10- to 15-year increase after cutting is followed by an extended period of debris-avalanche occurrence significantly below the rate observed in areas of older, established

vegetation usually used in determining a reference "natural" or undisturbed rate. If this is true, alteration of processes—such as surface erosion, creep, and root throw which slowly reload sites prone to fail by debris avalanching—may be the most significant effect of management on increases of soil erosion on the time scale of centuries.

CONCLUSIONS

Inventory of debris avalanches through a period of forest-management activity provides a means of assessing impacts on erosion. Results of inventories in steep forest land of the Pacific Rim indicate that clearcutting generally causes several-fold increases in debris-avalanche erosion over 10- to 20-year periods after cutting, and roads have a much higher impact in the relatively small area they affect. Very steep, highly dissected terrain has much higher frequency of debris-avalanche occurrence, but the small average volume of events results in soil transfer rates similar to other areas studied. A series of factors limit the usefulness of interpretations of inventory data on debris avalanches, but some of these problems can be overcome with improved data analysis.

ACKNOWLEDGMENTS

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TABLE 1. Frequency, average volume, and soil transfer rate of debris avalanches under forested, clearcut, and road right-of-way conditions in Mapleton land type 47, Siuslaw National Forest (Swanson et al., 1977; Ketcheson, 1978); H. J. Andrews Experimental Forest (Swanson and Dyrness, 1975); Alder Creek, Willamette National Forest, Oregon (Morrison, 1975); Stequaleho Creek, Olympic Peninsula, Washington (Fiksdal, 1974); selected drainages in the Coast Mountains, British Columbia (O'Loughlin, 1972, and personal communication); and Maimai, Westland, New Zealand (O'Loughlin and Gage, 1975 cited in O'Loughlin and Pearce, 1976). Shown in parentheses is the factor by which each value exceeds the comparable value for forest conditions.

Site	Period of record (yr)	Frequency (events km^{-2} yr^{-1})	Average volume (m^3)	Soil transfer rate (m^3 km^{-2} yr^{-1})
Forest				
Mapleton (land type 47)				
Swanson et al.	15	0.533	56	24
Ketcheson	15	0.432	25	11
H. J. Andrews	25	0.025	1,460	36
Alder Creek	25	0.023	1,990	45
Stequaleho Creek	84	0.015	4,660	72
Coast Mtns., B.C.	32	0.004	3,060	11
Maimai, N.Z. (estimated)		0.2	505	103
Clearcut				
Mapleton (land type 47)				
Swanson et al.	10	1.01 (1.9)	110 (2.0)	111 (4.6)
Ketcheson	15	1.50 (3.5)	34 (1.4)	128 (12)
H. J. Andrews	25	0.097 (3.9)	1,368 (0.9)	132 (3.7)
Alder Creek	15	0.267 (12)	440 (0.2)	117 (2.6)
Stequaleho Creek	6	0	—	—
Coast Mtns., B.C.	32	0.021 (5.2)	1,120 (0.4)	24 (2.2)
Maimai, N.Z.	6	4.1 (20)	560 (1.1)	2,260 (22)
Road right-of-way				
Mapleton (land type 47)				
Swanson et al.	15	8.23 (15)	425 (7.8)	3,500 (125)
H. J. Andrews	25	1.38 (55)	1,320 (0.9)	1,776 (49)
Alder Creek	15	8.33 (362)	1,870 (0.94)	15,600 (348)
Stequaleho Creek	6	19.4 (1293)	560 (0.12)	11,640 (160)
Coast Mtns., B.C.	32	0.26 (220)	3,320 (1.2)	262 (24)

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Watershed Management in the Coastal Ecosystem - Grants Pass Feb. 24, 1983

Outline: Maintenance of Site Productivity (Dick Miller)

Objective: To improve ability to predict the consequences of current forest management practices on site productivity.

Factors of site productivity:
= (f) stand, soil, microclimate, climate...

stand	soil	microclimate	climate
species	nutrient	aspect	ppt
density	moisture	slope	solar
regime	aeration	elevation	fog
rotation	stability		
	temperature		

Forest Practices: Affectable factors of site productivity:

Item	Visible effect on OM/nut.	Visible effect on soil phys.	soil nutrient	soil moisture	soil aeration	soil stability	soil temp.
Harvest							
Site prep.							
(Regeneration)							
Veg. control							
PCT, CT							
Fertilization							

Three truths:
1. Every action causes reaction(s)
2. Reaction(s) depend on initial conditions (where, when) and the action (what, how, extent). Corollary is we can't generalize about effects
3. Someone pays now ... or pays later

Bottom lines:
1. Quantifying long-term effects on stand productivity provides the direct and needed answer ... other quantifications provide substitute answers
2. Maintaining productivity - a conservative approach - is usually less costly and biologically more effective than restoration (Gessel 1981)

THE IMPACT OF EVEN-AGE FOREST MANAGEMENT ON PHYSICAL PROPERTIES OF SOILS

Henry A. Froehlich

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THE IMPACT OF EVEN-AGE FOREST MANAGEMENT ON PHYSICAL PROPERTIES OF SOILS

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Management of today's forests at almost any level of intensity implies the extensive use of machinery, not only for the initial harvest or stand conversion but for the application of a variety of silvicultural treatments as well. The equipment has the potential for markedly modifying the physical properties of the surface soil layers, which may significantly affect regeneration and future growth. Some form of crawler tractor or rubber-tired skidder is the primary machine used in all but the steepest regions of North American forests. Throughout the West, loggers commonly use crawler tractors or rubber-tired skidders on slopes up to 30 or 40 percent and occasionally on slopes up to 60 percent. Recently, the Forest Industries magazine reported on a rubber-tired skidder being used on slopes up to 84 percent. This is probably the exception, as most slopes above 40 percent supporting old-growth timber in the West have been harvested by some cable system, such as conventional high-lead or skyline methods. Until very recently, young-growth forests have been thinned almost exclusively by crawler tractors and skidders adapted from equipment used in harvesting old-growth timber. Now, loggers are beginning to develop cable systems for thinning and harvesting in young-growth stands.

The change in the properties of the surface soils as a result of harvesting and thinning may be either beneficial or harmful. Some degree of exposure of mineral soil is frequently desired as an aid to regeneration. Beyond the soil disturbance from scarification by dragged logs or machine movement, there is often a much deeper disturbance from the construction of tractor trails. This becomes especially visible as the slopes approach the maximum for the operation of ground-based

equipment. Another type of disturbance that is less visible is soil compaction in the routes used for machine and log movement. The areal extent of the surface disturbance and even the degree of compaction may be measured easily. The actual effect that this has on tree or stand growth over a rotation is not so readily observed and many unanswered questions remain on the subject.

FOREST SOIL CHARACTERISTICS

It is impossible within the scope of this paper to attempt to characterize the ideal soil system for each commercial forest species. Some things can be said that generally identify the highest producing sites, however. The significance of permeability, texture, and available moisture in influencing tree growth is recorded by research in forested areas around North America (1, 2, 4, 15, 35). Lutz (17) states that the critical apparent density varies with texture, and, "Optimum compactness is that apparent density at which the particular soil has the pore-size distribution that will give the water and air capacity and movement best suited for plant growth." The natural bulk density of high-site forest soils is generally quite low, normally increasing with depth. Expressed as bulk density, the surface layers of many productive soils are reported to have densities ranging from about 0.5 g/cc to 0.9 g/cc. Schlots, Lloyd, and Deardorff (29) provide detailed descriptions of 14 Douglas-fir soils in western Washington. The highest site-index values are on those soils with a porosity that permits free permeability of air and moisture throughout the profile. Forristall and Gessel (8) indicate that effective rooting depth is related to soil density and soil porosity and that the lower limit of the effective rooting depth corresponds to a marked increase in bulk density. Lutz' concept of an optimum compactness is supported by Steinbrenner's (31) findings on the relation of macroscopic pore space in the B horizon to site index of Douglas-fir in western Washington. He found that from a lower reading of 7 percent there is an increase in site with an increase in macroporosity up to 14 percent, followed by a decrease in site with an increase in pore space beyond this.

Thomas, Pomerening, and Simonson (38) provide a range of values for the particle size distribution for a large number of soil series found in Oregon as well as the site class for Douglas-fir commonly associated with these soil series. The most productive soils are found to be associated closely with a clay content of about 20 to 35 percent, with generally a wide range of particle sizes for the balance. For these soils to have the generally low bulk density and high porosity found under undisturbed conditions, a high degree of aggregation must be present. The loose, friable nature of these soils under natural conditions is the result of chemical and physical weathering and biological activity. Freezing-thawing, drying-wetting, root growth and decay, and the activity of micro- and macroscopic soil organisms have all played a part in forming the surface layers of soil into aggregates of mineral and organic material to provide a medium in which soil-water solutions may move relatively freely and adequate aeration may occur. Mechanical impedance of the coarse lateral and fine feeder roots under natural conditions is at a minimum. It is understandable that soils of this type would be readily subject to changes during logging.

THE COMPACTION PROCESS

A brief look at some of the equipment currently in use shows that many of the rubber-tired skidders weighing between 10,000 and 18,000 pounds have a rear-wheel ground pressure of from 20 to 28 lb/in^2. This would be static pressure at the rear wheel with the machine holding a normal turn of logs. Crawler-type tractors may range from about 10,000 to 36,000 lb or more, with ground pressures ranging from 6 to 13 lb/in^2 assuming equal distribution of weight over the whole track-bearing surface. Some logging equipment now under development is expected to have ground pressures of near 4 lb/in^2. These values of ground pressure are for static conditions of the fully equipped but unloaded tractors. The effect of the various sizes of loads, bouncing action, and vibration when the machine is in motion all add to this minimum, and the actual ground pressures may be increased several times.

Both vertical pressure and vibration change the physical condition of the soil. Any pressure on the soil surface greater than the natural internal friction will cause the aggregates to shift relative to each other and thus initially reduce the macroscopic pore space but basically retaining the aggregate units. With further pressure, plastic deformation will occur at contact points between the aggregates. Repeated trips over a soil surface also produce a kneading action that alternates the direction of pressure in a soil unit, thus accentuating compaction. Continued or repeated pressure and vibration in the presence of a sufficient moisture to lubricate shear planes within and between aggregates will cause puddled soils, a severe breakdown of the soil structure (20). Chancellor, Vomocil, and Aerf (3) provide a detailed sequence of soil change from compaction in their study of energy disposition in compression of agricultural soils.

A standard engineering test commonly used to determine the optimum moisture content for maximum soil compaction will help to visualize the influence of moisture in decreasing the shear strength. In this test, the selected soil is confined in a cylinder and subjected to a known compactive effort. The test is run at several increments of soil moisture with the results plotted as a curve from which the optimum moisture content may be read. An example of the type of curve produced by a clay loam forest soil is shown in Figure 1.

In this example, 95 percent of the maximum compaction occurs at a moisture content ranging from 14 to 30 percent. The absolute value of dry density would be somewhat higher were it not for the high organic matter content of the surface soils. Note that below the optimum moisture content, further compaction could be produced with a greater compactive effort. Above the optimum moisture level, the excess pore water pressure effectively limits the density that can be attained under the test conditions. Under unconfined conditions, however, the soil will become plastic enough to flow away from the compactive force with a rapid loss in soil structure. A soil with a wide range of particle sizes may be compacted to a greater density than more uniform soils and will compact at a lower

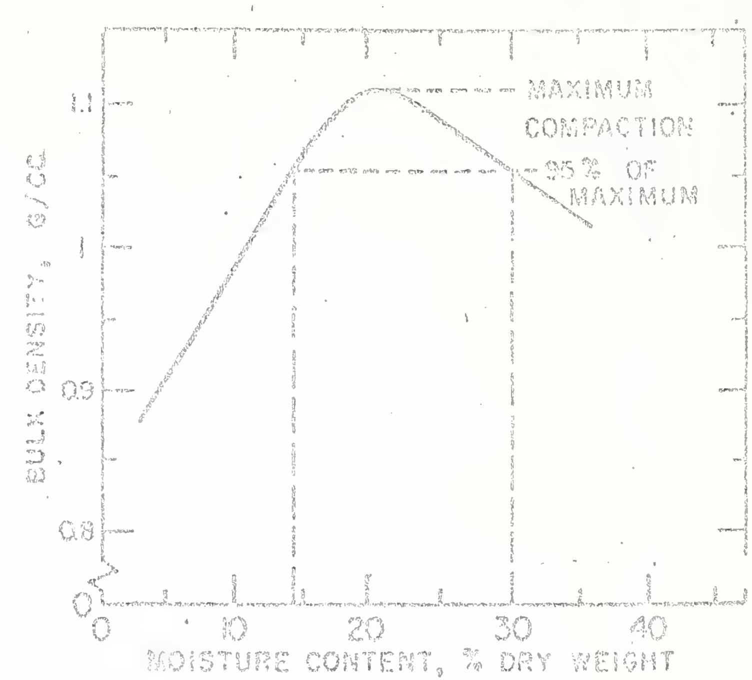

Figure 1. An example of the results from the Proctor Test of the surface layer of a clay loam forest soil.

moisture content. Sandy clays, for example, may be compacted to densities of 1.7 to 2.1 g/cc at only 8- to 15-percent soil moisture, but the maximum densities of clays are in the range of 1.5 to 1.7 g/cc achieved at 20- to 30-percent soil moisture. Of course, these maximum densities seldom are achieved under field conditions. The Corps of Engineers, Waterways Experiment Station (40), reports that this optimum moisture content for compaction occurs about midway between soil moisture tensions of 0.06 and 15 atmospheres, approximating field capacity and wilting percentage. This general rule appears to be realistic for soils ranging from clays to sandy loams.

The degree of soil compaction actually achieved is thus strongly related to the moisture content, degree of aggregation and organic content. Day and Holmgren (5) aptly described the soil reaction to pressure:

During compression the forces exerted upon an individual soil aggregate by the surrounding aggregates comprise a complicated force system. The factors which determine the shearing strength are the internal cohesive forces and the friction forces. Plastic deformation can be expected only if the applied stress is sufficiently great to overcome the shearing strength; otherwise, the aggregate will function as a mechanically stable structural unit. This principle is a necessary starting point in the interpretation of soil compression.

These authors also demonstrate the effect of changes of moisture content and three levels of pressure on bulk density of a silty clay loam (Table 1). Thus at about 25-percent moisture content, a pressure of 14 lb/in^2 increased bulk density by 64 percent, but, at about 15-percent moisture, a comparable pressure only increased bulk density by 27 percent.

Table 1. Relation of Moisture Content and Pressure to the Bulk Density of a Silty Loam (5).

Water content	Initial	Final
%	G/cc	G/cc
7 lb/in.^{2}		
25.3	0.73	1.23
15.4	0.78	0.93
14 lb/in.^{2}		
25.4	0.81	1.33
15.3	0.78	0.99
21 lb/in.^{2}		
25.4	0.84	1.69
15.5	0.78	1.07

PHYSICAL CHANGES IN SOIL PRODUCED BY HARVESTING AND THINNING

Density, Porosity, and Infiltration

It is impossible to make a direct comparison of the laboratory compaction tests to the compaction produced in the field. That is, the energy applied in the Proctor test may not be duplicated by the tire or tread of a harvesting machine. The test should serve as a good index of what can occur under field conditions, however. Steinbrenner and Gessel (34) observed a 35-percent increase in bulk density on tractor skidroads after six trips with a tractor under winter conditions on a silty clay loam (Olympic series). Soil density increased about 17 percent after six trips under summer conditions, and it required about seven or eight trips to achieve a 35-percent increase in density under summer conditions. Weaver and Jamison (41) indicate that the first four passes of a tractor produced the greatest increase in density in the agricultural soils of their study. My observations on tractor thinning of areas on a reddish brown clay (Nekia series) indicate a 25-percent increase in bulk density between the logged areas and the heavily disturbed areas. The change in density is related almost directly to the loss in macroscopic pore space, and this loss in pore space markedly influences permeability and infiltration.

Steinbrenner and Gessel (33) found that tractor-cutover land had a 2.4-percent increase in bulk density, a 35-percent loss in permeability, and a 10-percent decrease in macroscopic pore space compared to undisturbed soils. Skid trails in the same area showed a 53-percent loss in macroscopic pore space and a 93-percent loss in permeability. Steinbrenner (30) found that, although it took six trips with the tractor to reach maximum compaction, nearly all of the loss in infiltration rates occured with the first two trips. It required about four trips in summer to achieve the same loss in infiltration. Dyrness (6) noted that undisturbed forest soils in his study on the Andrews Experimental Forest in the Oregon Cascades had about 77-percent pore space, but compacted soils in adjacent tractor logging averaged about 63 percent. This change in pore space

was brought about by a change in bulk density from 0.603 g/cc in the undisturbed areas to 0.975 g/cc in the compacted areas of tractor logging.

Perry (27) studied the soil conditions under loblolly pine 26 years after planting. He found that infiltration for a given quantity of water required 3.5 min in planted fields and 18.5 min in areas compacted at the time of planting. On currently compacted sites the infiltration time ranged from 80 min to more than 4 hr. Trimble and Weitzman (39) found that it took 619 times longer for a given quantity of water to enter the soil of a skidroad than to enter the A-horizon of an undisturbed forest soil, and 20 times longer than to enter into the B-horizon. Munns (23) found that infiltration was reduced by 75 percent on tractor-logged sites in the pine region of California, and Hulisashvili (14) observed that after clearcutting of a pine-spruce-oak forest on brown forest soils of the eastern Georgia regions of the Soviet Union there was about 50 percent less noncapillary pore space. This loss in large pore space gave rise to a 3.5-fold decrease in infiltration.

Depth of Compaction

The depth to which the compaction is found is quite variable. It penetrates deeper under wet soil conditions than under dry, and the more porous the soil initially the greater the compaction depth. Munns (23) found that compaction under tractor logging with relatively light equipment in the pine region of California penetrated to at least 10 in. in depth. Lull (16) reviewed a number of studies that show compaction depths from 6 to 24 in. He concludes that the depth to which compaction occurs does not generally extend beyond 12 in. below the bearing surface and that laterally the effect is limited to 12 to 18 in. from the tire or tread. It is difficult to relate compaction on agricultural soils to that occurring in logging, because the latter is produced by a combination of tire or tread pressure, kneading action, and vibration from the equipment plus the pressure and scarification from the turn of logs.

less reduction of pore space

Areal Extent of Soil Disturbance

As early as 1947, Munns (23) raised the question of possible damage to soils when he observed that tractor trails and roads covered from 25 to 40 percent of a logged area in the pine region of California. Steinbrenner (30) reported that 26 percent of a tractor-logged area was occupied by tractor skidroads and included deep displacement of soil, surface compaction, and puddling of saturated soils. Dyrness (6) also found that although up to two-thirds of a tractor-logged area may be disturbed during logging, about 26 percent of the area may be classified as compacted. His report also shows that high-lead cable logging during clearcutting produced about 9-percent surface soil compaction.

Wooldridge (42) compared the soil disturbance produced by partial cutting with tractor logging and a skyline crane system. He reports that deep soil disturbance occurs on about 16 percent of the tractor logging and only 3.2 percent under the skyline crane method. Haupt (13) observed that partial cutting of old-growth ponderosa pine produced soil disturbance ranging from 3 to 18 percent, varying with the initial stand volume, volume removed, and size of equipment. My observations of several thinnings in 50- to 90-year-old Douglas-fir lead me to expect that from 40 to 60 percent of the thinned sites will be disturbed, with from 20 to 30 percent of the area having heavy disturbance and compaction.

EFFECT ON SEEDLING AND TREE GROWTH

The literature on the effect of compaction on agricultural crops is quite extensive, but that dealing with forest crops is limited. The effect of compaction on tree roots has centered around three factors; physical impedence, gaseous exchange, and nutrient and moisture movement (28). Zimmerman and Kardos (44) found that root penetration is correlated negatively with bulk density, but that there is a wide difference between plant species in ability to penetrate dense soils. Forristall and Gessel (8) noted that in Alderwood gravelly loam, Douglas-fir

and western hemlock root growth is restricted significantly when the bulk density approaches 1.25 g/cc. Minore, Smith, and Woollard (21) observed the effects of high soil density on seedling root growth of seven northwestern tree species. Soil columns were compacted to 1.32, 1.45, and 1.59 g/cc and seedlings allowed to grow for 2 years under greenhouse conditions. The roots of all seven species grew through the 1.32-density soil cores. The roots of western redcedar, Sitka spruce, and western hemlock did not penetrate 1.45-density cores, but roots of red alder, lodgepole pine, and Douglas-fir grew through them. No roots penetrated the 1.59-density cores. Steinbrenner (31) observed that no roots were found in soil layers that preclude passage of air at 15 lb/in². Pearson and Marsh (25) found that the compaction of certain soils in the Black Hills region had an unfavorable effect on the reproduction of ponderosa pine because the altered soil structure impeded root development. The undesirable characteristics were accentuated by logging operations in wet weather. Steinbrenner (32) noted that plants on the skid trails tended to have shallow root systems brought about by the reduction in natural macroscopic pore space, lower permeability, and reduced oxygen capacity of the soil. Sutton (36) reviewed a number of papers on the form and development of conifer root systems and concluded that roots will grow only in that part of the soil where moisture, aeration, and mechanical properties are favorable. Aeration has been shown to influence total root surface area, size of root-hair zone, and fibrousness in roots of several species. Pearse (26), working with Douglas-fir and western hemlock grown in sandy loam compacted to densities of 0.59, 0.84 and 1.02 g/cc, found that total root length after 8 weeks was least in the most compact soil.

Hatchell, Ralston, and Foil (11), working with loblolly pine, also found that root weight was correlated negatively with bulk density over a range in density from 0.8 to 1.4 g/cc. Shoot-root ratio was correlated positively with noncapillary porosity and oxygen diffusion rate. Foil and Ralston (7) reported that compaction of a variety of soils, whether at 3.5 or 10.[illegible] kg/cm² of surface pressure, greatly reduced size and weight of loblolly pine seedlings. Small differences in growth among compaction treatments indicated that even the smallest pressure applied reduced soil aeration and increased mechanical impedance to root growth to unfavorable levels. Some height growth effects were observed on first-year growth of loblolly pine. On one area, there was a small but statistically nonsignificant height-growth difference between seedlings on a secondary skid trail and an undisturbed area. On a second area, however, seedlings averaged 3.8 in. on a primary skid trail compared to 4.8 in. on an undisturbed site. On primary skid trails there was a 43-percent loss in seedling height growth on the first site and a 53-percent loss on the second site.

Youngberg (43) measured the influence of soil conditions after tractor logging on planted Douglas-fir seedlings. He showed that the average leader growth in the cutover area was 6.8 in., compared to 5.3 in. for seedlings growing on a tractor trail berm and 3.9 in. for seedlings growing on a heavily disturbed tractor trail, a 43-percent reduction from those in the cutover area. The Bureau of Land Management, Eugene District, reported a 57-percent loss in terminal leader growth between undisturbed sites and tractor skidroads on 8-year-old Douglas-fir seedlings.[1] Their data also indicate the relation of bulk density to leader growth (Table 2).

Table 2. The Effect of Bulk Density on Leader Growth[1].

Bulk density	Leader growth
G/cc	Inches
0.84	26.69 ± 5.95[2]
0.94	23.71 ± 4.62
1.06	22.88 ± 4.70
1.15	18.60 ± 6.39
1.24	17.22 ± 7.79

[1]From unpublished paper on file at Bureau of Land Management, Eugene District, Eugene, Oregon.
[2]Plus or minus standard error of the mean.

Actual values for effects on residual tree growth seldom have been reported. Olson (24) described a "logging shock" that results in considerable loss of trees and arrested growth in residual stands of western white pine. Perry (27) measured growth on loblolly pine 26 years after they were planted. Data from 30 pairs of trees showed a significant reduction in growth between trees growing on compacted areas and those in surrounding fields (Table 3).

Table 3. Average Growth of Trees in Compacted and Uncompacted Areas (27).

Location	Dbh	Height	Volume
	In.	Ft	Cu ft
Woods road	6.3	54	4.1
Surrounding fields	8.7	62	8.8

In an effort to determine the combined effect of tractor and load on soils and tree growth, Moehring and Rawls (22) selected a series of loblolly pines in a 40-year-old stand. The soil was compacted by a small tractor pulling a load of three 10-foot logs passing six times near the tree. The treatment was applied during wet weather, and the results were measured in terms of growth 5 years later. Growth was found to be affected only slightly when compaction was limited to one side of a tree. Compaction on two, three, and four sides of a tree produced 13.7, 36.3, and 43.4 percent less volume over the 5-year period.

PERSISTENCE OF COMPACTED CONDITION

The longevity of the compacted condition is of major interest to forest managers, but data on rates of recovery still are limited. Of course, the same forces that were instrumental in bringing about the porous soil structure and aggregation will begin to restore the soils to their natural tilth. The process

sometimes may be quite slow, and possibly some threshold or limit exists from which a given soil may recover quickly and beyond which a long interval will be required. Coarse-textured soils apparently will recover more quickly than fine-textured soils. Mace reports on the recovery of a sandy soil after thinning of a 90- to 100-year-old red pine stand in Minnesota. Immediately after thinning, a 5-percent increase in bulk density was determined for the skidder trails used in a tree-length yarding, and an 11-percent increase in bulk density was found in skidder tracks used in skidding full trees. The lightly compacted soils were markedly recovered after one overwintering period. The somewhat heavier compaction produced by the full-tree skidding is recovering at a slower rate. In both, the compaction near 6-inch depth had recovered significantly in 1 year, apparently because freezing and thawing are more prevalent at lower depths in sandy soils and permit a quicker recovery (18, 19).

In western forests, soil freezing appears to be much less prevalent. Hale (9, 10) notes that the forest soils of the high Cascade Mountains, both east and west side, remained unfrozen throughout the winter and spring of 1949-1950. In the following winter, frost occasionally penetrated to 3-inch depth under lodgepole pine stands and about 2 inches under ponderosa pine stands. At no time during the observation were there extensive areas of impermeable frozen ground. In the Coast Range of Oregon and Washington freezing-thawing cycles apprently will not add significantly to natural recovery of soil tilth.

Munns (23) noted little or no change in the infiltration rate within 2 years after logging. Tackle (37) also used infiltration as an index of soil condition on a silty clay loam after logging on western larch and Douglas-fir land in the intermountain region. The relative infiltration values in the first and fifth years after logging are shown in Table 4. The scarified condition listed above indicates that the litter layer was removed or disturbed excessively by the tractor in logging or piling slash. Lutz (17) cites an example of soil compaction from tractor tire pressure in orange groves and noted that the

Table 4. Infiltration Values in the First and Fifth Years After Logging (37).

Condition	1st year	5th year
	%	%
Undisturbed	100	100
Scarified	15.4	48
Broadcast burn	62.5	135[1]
Tractor skidroad	4.1	4.1

[1]Had recovered by 3-4 years.

elimination of cultivation for 8 years and the growing of cover crops partly restored a good physical condition.

AMELIORATION OF COMPACTED CONDITION

Artificial means of loosening the compacted soils may be possible, but the results of such efforts are highly variable. Both the Bureau of Land Management and U. S. Forest Service have required certain compacted areas to be scarified, and Steinbrenner (32) reports that this also has been done on some private forest lands. The success of the scarification projects is not known at this time, but some measure of improvement apparently is achieved. Experience from agricultural soils indicates that the condition of the soil at the time of scarification is crucial and that if the work is done under a moisture content near the optimum for compaction, further compaction may result. Hatchell (12) studied the effect of compaction and loosening treatments on loblolly pine grown in cores of compacted soil and noncompacted soil taken from adjacent sites. A randomly selected portion of each soil condition was loosened to 0-, 3- and 6-in. depths. Loosening was achieved by removing the core and breaking it into particles no l[illegible]r than 1/4 in. in diameter and then loosely packing the soil into a soil column. The soils in their compacted condition

had bulk densities from 1.12 g/cc for a clay loam to 1.72 for a silt loam. The author makes a special note that the greatest retardation in growth on compacted soil as compared to noncompacted soil was observed on silt loam soil. Loosening compacted soils did not produce a significant increase in growth of shoots and roots. Mean dry shoot weights for compacted soils loosened to 0-, 3-, and 6-in. depths were respectively 164, 211, and 207 mg. and respective root weights were 66, 79, and 74 mg; these weights are considerably less than those on noncompacted soils with no loosening—344 mg for shoots and 105 mg for roots. Loosening compacted cores resulted in a percentage of seedling establishment that was almost as high as the percentage on cores of normal soil, however. Longer term studies clearly are needed to learn what effect various loosening or scarification treatments have on restoring desirable soil properties. Mechanical loosening, if done under proper conditions, apparently will at least make it possible for natural forces to act more quickly on the compacted soils.

Some tree species and many other grass and shrub species can penetrate dense soil to a greater extent than some of our desirable commercial species. Perhaps alder, *Ceanothus*, or other pioneer species will serve a useful purpose in this fashion in the Northwest. Lull (16) reviews papers that indicate the beneficial role of mustard, grasses, and legumes in improving infiltration rates. The increasing practice of seeding skid trails and landings with grass species as an erosion-control measure also should be beneficial in speeding the recovery of the compacted soil.

CONCLUSION

The problem of the effect of the harvesting and thinning processes on desirable physical soil properties varies markedly from soil to soil, but at times it can cause a serious loss in seedling establishment, growth, and long-term productivity. Some degree of soil disturbance and compaction is almost inevitable in harvesting, but it may be reduced significantly by the proper selection of equipment and the timing of its use. The combination of a high degree of mechanization utili[illegible]g heavy

equipment and the naturally porous nature of the most productive soils intensifies the problem. An understanding of the compaction process and the physical nature of the soils we manage will aid in reducing the degree and amount of soil problems from intensive use of machines. The actual physical change is primarily one of reduction of noncapillary pore space, which in turn reduces infiltration, permeability, and gaseous exchange. A smaller, but at times significant, area may be puddled, which represents a severe loss of soil structure. Various studies in old-growth harvesting indicate that from one-quarter to one-third of a clearcut area may be affected seriously by ground-based machines and from 3 to 9 percent of the area may be affected by cable harvesting systems.

Some studies have shown that the effect of compaction on seedling growth varies greatly between soils and with tree species. There is a consistent trend toward a reduction of both seedling root length and shoot or leader growth. Losses in height growth range from 14 to 53 percent depending on soil type, degree of compaction, and tree species. The trend of growth on residual trees affected by compaction also is correlated negatively with density and may range from relatively small reduction to over 40-percent loss in growth on individual trees.

The compaction under some conditions persists for a long time, perhaps at least a few decades and is more persistent in clayey soils than coarse-textured soils. Freezing and thawing and drying and wetting appear to be major agents in restoring soil structure that act most rapidly on the least compacted soils. Mechanical means of loosening compacted soils appear to hold some promise under some conditions but primarily as a means of aiding natural forces to act on the soil mass. Biological means of improving soil conditions are probably more certain, and the use of selected plant species to pioneer the recovery of the affected soil shows considerable promise.

A major question still to be answered is: What is the effect of the disturbance and compaction on a forest stand over a rotation? It may be that those trees in the vicinity of compacted sites may be able to compensate for some loss of

rooting space and the release from competition from the disturbed area may still allow accelerated growth. Seedlings or planted stock on the skid trails clearly appear to suffer growth losses, but it has been suggested that those plants adjacent to compacted areas may not suffer serious growth loss if the soil condition is ameliorated by the time the root development of the young stand requires the use of these areas. Because the potential for growth loss appears to be substantial, however, especially under intensive forest management, every effort to minimize the impact is warranted.

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A COMPARISON OF HARVESTING METHODS AND THEIR IMPACT ON SOILS AND ENVIRONMENT IN THE PACIFIC NORTHWEST

K. Cromack, Jr., F. J. Swanson and C. C. Grier[1]

The productive evergreen coniferous forests of the Pacific Northwest have accumulated large biomass in individual stands. These evergreen forests are adapted to a winter-wet, summer-dry environment enabling photosynthesis and nutrient uptake and storage during the relatively mild winters (Waring & Franklin 1979). Pacific Northwest forests and their associated streams also support important fisheries resources. Possible future constraints in energy supply, coupled with the need to maintain long-term site productivity, necessitate continued improvement in forest management methods (Bengtson 1978; Bormann & Likens 1979).

To evaluate management activities upon forest ecosystems one must include terrestrial and aquatic habitats and the materials transferred by nutrient cycling and erosion processes. Although large living trees are the dominant element of the mature forest ecosystem, recent studies show other parts of the ecosystem also play important roles. Other functional components of Pacific Northwest coniferous forests, such as the substantial leaf biomass and leaf area of mature conifer stands (Waring & Franklin 1979); large accumulations of roots (Santantonio et al. 1977); dominance of mycorrhizae in belowground organic matter inputs (Fogel & Hunt 1979); and relatively deep and fertile soils from which N leaching losses are small (Sollins et al, in press) all contribute to forest productivity.

In forested watersheds the aquatic environment is small streams. Such streams comprise 86% of the total length of river channels in the US (Leopold, Wolman & Miller 1964). The physical character and energy base of these small

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streams are dominated by adjacent forests which supply energy, minerals and shade, thus regulating the rate of stream production (Meehan, Swanson & Sedell 1977). Much organic and inorganic material is transferred from the forest to the streams by biotic and physical processes. The storage or transfer of these materials also represents nutrient capital. A diagram illustrates the sources and fates of material coming into and passing through a forest ecosystem (Fig. 1). By careful budgets of where materials are stored and how rapidly they are accumulated or lost, long-term responses to various management options may be evaluated.

As in the case of nutrient cycling regimes, soil/sediment movement through ecosystems is viewed as a series of storage sites linked by transfer processes (Fig. 2). Soil is moved down hillslopes into channels and then downstream. This material transfer, or erosion, is accomplished by a complex set of interacting processes (Dietrich & Dunne 1978, Swanson et al. in press). Soil and sediment are stored at a variety of sites on hillslopes, in channels, and at the interface between slope and channel. Both the capacity of storage sites and rates of transfer processes on slopes and in small streams are regulated by live and dead vegetation. Root systems, for example, anchor the soil mass and reduce potential for shallow, rapid soil mass movements (debris avalanches). Logs in streams temporarily trap sediment and dissipate energy of stream water, thereby reducing its sediment transport capability. To assess management impacts on soil loss and sedimentation one must understand this soil/sediment routing system and identify major processes and storage sites on land and in streams.

Leaching and erosion losses affect one another in both terrestrial and aquatic systems. For example, erosion affects the stability of substrates and in turn the leaching of nutrients in both terrestrial and aquatic environments. Likewise, organic matter in both living and dead forms regulates the erosion rate. In the western US, where coniferous forests dominate, the harvesting of these forests has a particularly large impact upon the aquatic environment.

HARVEST EFFECTS

Current silvicultural and harvesting techniques in the Pacific Northwest attempt to minimize cost in adapting to rugged and unstable terrain. A typical harvest operation requires: roads and landing; felling, bucking and hauling of logs; site preparation; and planting or reseeding of the site. Until the time of final harvest the stand may be entered many times to protect it from fire, insects or disease. To increase growth, fertilization, thinning and brush control are often practical. Although management practices are quite varied we will attempt to generalize the kind and extent of impact upon erosion and leaching.

Effects of Harvesting Upon Subsequent Vegetation

Cutting a forest has a variety of impacts on vegetation. Logging in effect is a perturbation that initiates secondary ecological succession. The vegetation response to perturbation depends upon the type of logging, environmental conditions, local patterns of plant succession and the success of

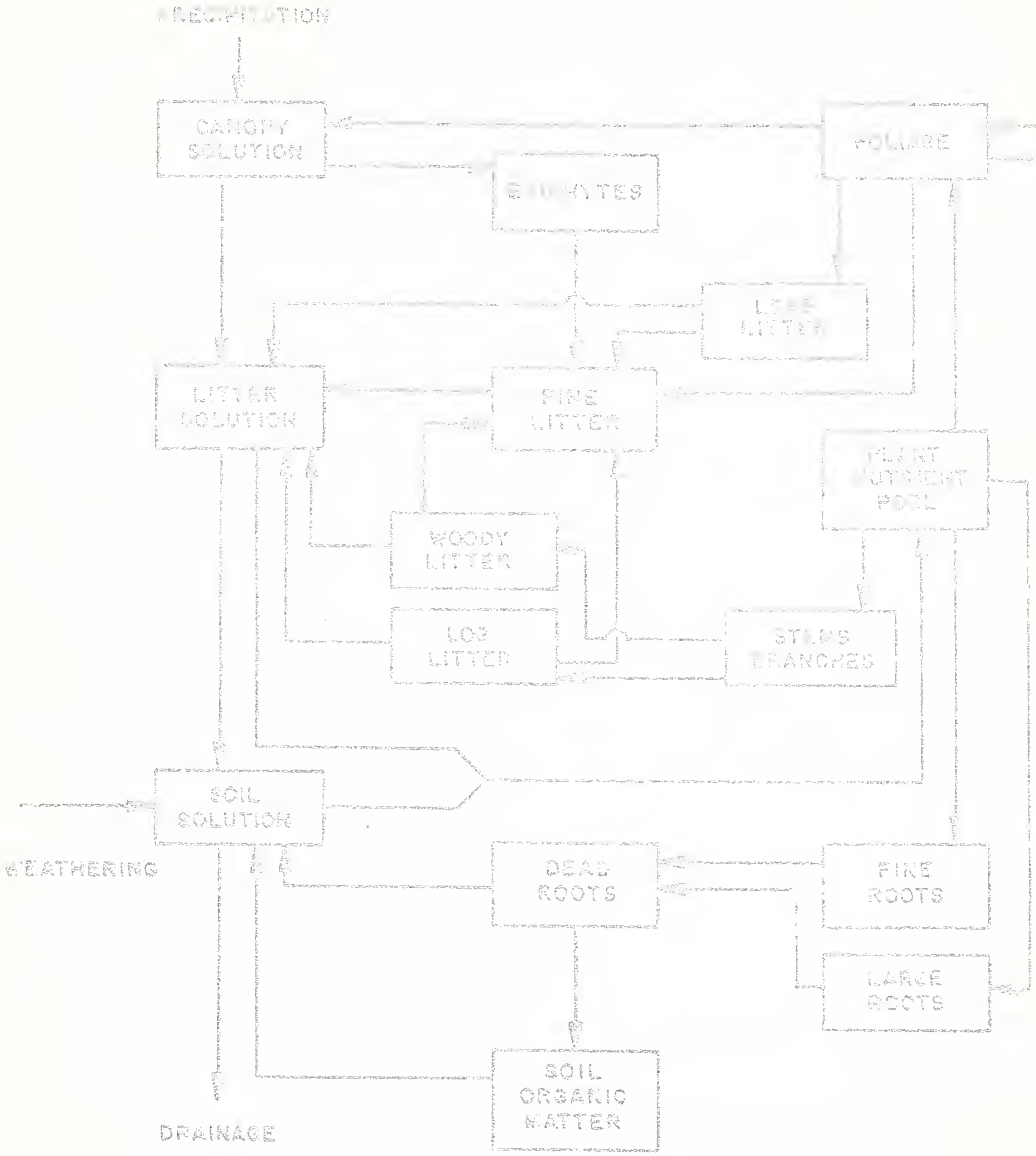

Figure 1. Diagram of a coniferous forest ecosystem showing functional components and nutrient transfers (after Sollins et al., in press).

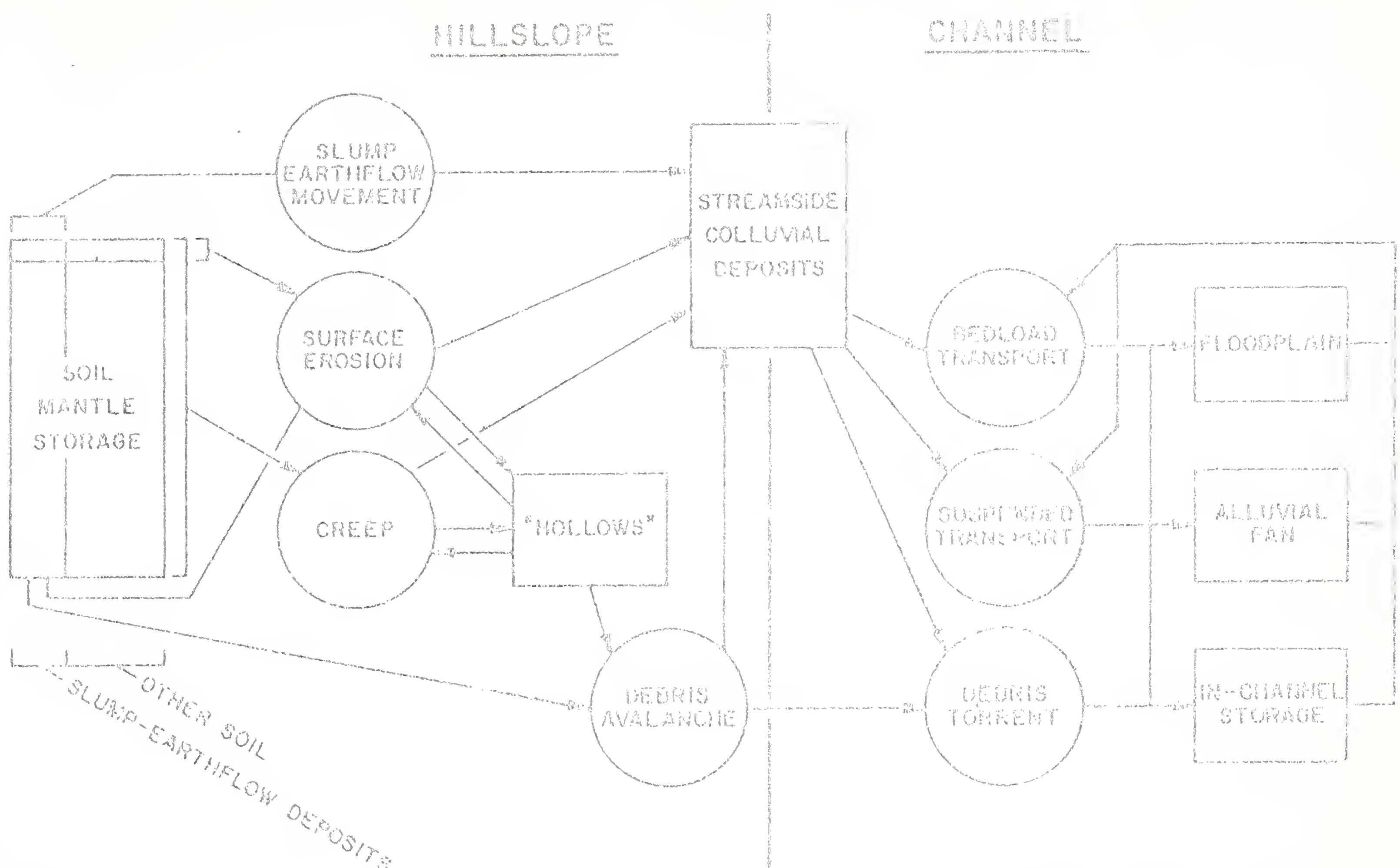

Figure 2. Flow chart showing soil and sediment storage sites (boxes) and transfer processes (circles). Debris avalanches are shallow, rapid soil mass movements from slopes. Slump-earthflow is slow, deep-seated soil movement on a discrete failure surface. Creep is subtle soil mantle deformation without development of discrete failure surface. Surface erosion includes sheetwash, rill, dry period and freeze-thaw related processes. Debris torrents are rapid movements of debris down channels. "Hollows" occur at headward tips of channels (Dietrich & Dunne 1978). In-channel storage occurs in gravel bars and is associated with organic debris accumulations. See Dietrich and Dunne (1978) and Swanson et al. (in press) for additional discussion and definitions.

regenerating efforts. Changes in the vegetation immediately following harvest are an important factor to be considered in stand regeneration. Release of brushy species by logging can result in competition for young conifers.

Methods of shrub treatment will influence brush development. When a site is left unburned, species already present will dominate. For example, if salal (*Gaultheria shallon*) or vine maple (*Acer circinatum*) were present in the understory prior to harvest, they will be following harvest.

Slash burning, on the other hand, favors new species absent from the original understory. Species that reproduce by windborne seed, and germination from buried seed appear to be favored by slash burning (Ahlgren & Ahlgren 1960).

Three stages in secondary succession should be recognized (Ahlgren & Ahlgren 1960, Franklin & Dyrness 1973): 1) herbaceous stage, 2) shrub stage; and 3) young forest stage. Other stages are absent in managed forests.

On sites that are broadcast burned, succession generally proceeds through all three stages, but on unburned sites, the herbaceous stage is not always present. In areas where logging exposes mineral soil, patterns of secondary succession resemble those on burned sites (Dyrness 1973).

1)The herbaceous stage occupies the site for the first few years and consists mainly of invading herbaceous species such as *Senecio sylvaticus* and fireweed (*Epilobium angustifolium*). Annuals such as *Senecio* are among the first to invade a logged site. These are replaced later by perennials such as fireweed and bracken fern (*Pteridium aquilinum*) that dominate the site for two or three years. Species invading recently disturbed sites tend to be "generalists" having rather wide ecological tolerances (Franklin & Dyrness 1973). Eventually shrubs become established and replace the herbs.

2)The shrub stage of succession is dominated by species such as salal, vine maple or snowbrush (*Ceanothus velutinus*). In contrast to the wider ranging generalists in the herbaceous stage, the shrub species are more closely adapted to soils and climate of the site. Thus on drier sites grow species such as salal and hazel (*Corylus cornuta var. californica*). On moderately moist sites, salal may still be found with *Rhododendron macrophyllum*, Oregon grape (*Berberis nervosa*), vine maple, and snowbrush. Snowbrush is a common shrub on burned sites in the Oregon Cascades. Seeds of this species lie dormant in the soil until a fire triggers germination. Consequently, snowbrush is common on clearcut areas but usually is absent from established forests. In western Washington, salal, vine maple and Oregon grape are the primary shrub species occupying more moist harvested sites. On wetter sites the most common species are red alder (*Alnus rubra*), salmonberry (*Rubus spectabilis*) and swordfern (*Polystichum munitum*).

3) The time required for a forest to develop varies. The third stage of succession, development of a seral forest stand, is dependent on factors such as the type of disturbance, means of establishment (e.g. planting, natural seeding, advanced regeneration) and environmental conditions. Probably the most common occurrence, especially with the advent of large scale tree planting, is the development of a pure, even-aged stand of Douglas-fir. These stands eventually become dense enough to eliminate almost all of the competing

understory and, barring disturbance, remain this way until harvest. In wetter, cooler areas of the Northwest, such as the Oregon and Washington coasts or the northern Cascades of Washington, dense, young stands of western hemlock become established following the brush stage (Franklin & Dyrness 1973). This frequently occurs in spite of initial planting with Douglas-fir seedlings.

Red alder is among the first trees to re-establish on many of the moist and wetter sites. Pure stands of red alder are common in the Oregon and Washington coastal mountains and in many parts of the Puget Sound region. Formerly a common species only in riparian habitats, red alder has substantially expanded its area of occupancy - mainly as a result of earlier logging and fires. During normal succession, alder gradually is replaced by Douglas-fir or western hemlock. As alder can dominate a site for up to 90 years (Franklin & Dyrness 1973), foresters apply herbicides or harvest the alder to speed conversion to the more valuable conifers.

Effect of Harvesting on Nutrient Losses

Nutrient and erosion losses from undisturbed forest ecosystems are relatively small for elements such as N, P, and K (Sollins et al. in press). For elements such as Ca, Mg and Na, stream export from deep weathering is considerably greater (Fredriksen 1972; Sollins et al. in press). They are, however, accelerated following major disturbances such as logging, fire or major storms (Cole & Gessel 1965; Fredriksen 1971). In the Pacific Northwest, when watersheds have been gauged and sediment trapped, net losses of various elements are known (Fredriksen 1971, 1972 & Fredriksen[2]). Although accelerated nutrient losses of N, P, K, Ca and Mg do occur following logging (Cole & Gessel 1955; Fredriksen 1971), more concern exists for loss of N since it generally is a limiting element in Pacific Northwest forests.

Nutrient cycling studies in a variety of Pacific Northwest forests report N inputs in precipitation ranging from 1.1-4.1 kg/ha/yr; organic N in dust may add an additional 0.06-1.3 kg/ha/yr (Fredriksen[2]). In marked contrast, N in precipitation is as high as 21 kg/ha/yr in northeastern forests at Hubbard Brook (Bormann, Likens & Melillo 1977), reflecting urban and industrial pollution N inputs to the atmosphere. At the Coweeta Hydrologic Laboratory in North Carolina, Swank and Douglas (1977) reported precipitation contained N totalling 4.5-7.2 kg/ha/yr.

Under relatively undisturbed conditions, N precipitation and dryfall inputs generally exceed system losses so that net accumulation of N may be occurring (Fredriksen[2]). Such short-term nutrient accumulation must be balanced, however, against episodic losses of soil and litter during major erosion events (Fredriksen 1970), wildfire, harvesting and slash burning.

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[2] R. L. Fredriksen. 1975. Nitrogen, phosphorus, and particulate matter budgets of five coniferous forest ecosystems in the western Cascades Range, Oregon. Ph.D. thesis. Oregon State University, Corvallis, Oregon.

Soil disturbance and compaction, and fire-induced hydrophobic behavior of surface soil are two of the most important in situ alterations of forest soils. Compaction and disturbance occur during roading and harvest activities. Froehlich (1973a, 1974, 1976, 1978) provides excellent summaries of research on these topics for the Pacific Northwest. The portion of a management unit compacted or otherwise disturbed varies greatly with the types of road and harvest systems employed (Table 1). Of course, terrain, equipment capabilities, definitions of ground surface conditions and other factors vary somewhat from study to study, but several general relations between the degree of soil disturbance and logging systems do emerge.

In general, the most complex, expensive yarding systems are designed to operate in the steepest terrain and to have the least impact in terms of soil compaction and disturbance. Clearcutting by tractor yarding generally occurs on slopes less than 50%. In tractor yarded units, values of percent of area compacted and deeply disturbed range from about 20 to 40% (Table 1 and other studies). Dyrness (1965, 1967a, 1972) reports 36% of a tractor yarded clearcut as deeply disturbed or compacted, while skyline and balloon yarded units had only 8 and 4%, of the total area in these two disturbance classes. Results of other studies also show less soil disturbance by yarding systems capable of greater suspension of logs.

Proportion of selectively cut areas disturbed and compacted by tractor and skyline methods are remarkably similar to areas affected by these systems in clearcuts (Table 1 and other studies). This results from the need for access to all areas of a harvest unit regardless of whether it is being thinned or clearcut.

Exposure of bare mineral soil by yarding or site preparation activities often is desirable to aid establishment of species such as Douglas-fir, but soil compaction may significantly reduce site productivity. The persistence and long-term impact of compaction depends on severity of initial treatment, the ability of various species to cope with compacted soils, and rates of processes tending to decompact the soil. Youngberg (1959) reports a 43% reduction in seedling height growth on primary skid trails relative to uncompacted clearcut areas. The greatest reported impact of compaction on growth of Douglas-fir is in unpublished Bureau of Land Management data (Froehlich 1973a) which indicate a 57% reduction in height growth of 8-year-old trees relative to uncompacted sites in clearcuts. On a 55-year-old railroad yarded clearcut Power (1974) estimates a 40% reduction of yield from compacted areas.

Rate of recovery of compacted areas in the Pacific Northwest has received very little study, but is known to vary greatly with soil type and effectiveness of decompacting processes such as freeze-thaw action (Froehlich 1973a). In much of the Douglas-fir region processes of soil decompaction operate slowly or infrequently, so in many areas compacted conditions may persist for years.

Development of hydrophobic soil conditions in the Pacific Northwest has been observed following wildfire (Dyrness 1976) and slash burning (J. Maxwell, personal communication). Dyrness (1976) observed overland flow and erosion due to decreased infiltration rates from volcanic ash soils after a wildfire

through a predominantly *Pinus contorta* forest. Conditions of increased water repellency of soil at depths up to 23 cm lasted for five years. Effects of slash fires on soil wettability and related accelerated erosion have not been researched in the western coniferous forest. Many forest soils in the region are coarse-textured and occur on steep slopes, conditions which form soils with high infiltration rates even after fire, and prone to surface erosion by dry ravel rather than by sheet and rill erosion. Increases in peak flows from clearcut and burned watersheds in western Oregon have been interpreted in terms of road density and soil compaction rather than hydrophobic soils (Harr 1976). Therefore, development of hydrophobic soils by slash fires has not been considered a significant problem in the Pacific Northwest.

Effects of forest management activities on soil erosion have been reviewed by Rice et al. (1971), Megahan (1972), Stone (1973), Fredriksen et al. (1975), Swanston and Swanson (1976) and others. Most studies of soil erosion from forest land have been based on sediment yield from small watersheds, although some plot studies (Hauge 1977) and process-level investigations (Dyrness 1967b, Swanson & Dyrness 1975, and others) assess management effects. The magnitude of increases in soil erosion due to management activities is highly variable and depends largely on relationships between the type of harvest sequence used and the susceptibility of the landscape to erosion. A particular activity may have very different impacts under differing soil and slope conditions. Careful practices in difficult, erosion-prone terrain may have soil erosion impacts comparable to poor practices carried out on naturally stable ground.

Several examples from among the experimental watershed studies in western Oregon demonstrate the broad range of watershed responses to harvest activities. In a case of minimal impact of harvest activities, the Fox Creek watersheds in the Bull Run drainage, western Cascade Mts., experienced no detectable increases in suspended sediment export following 25% clearcutting of a small, gently sloping watershed with soils of low erosion potential on a 71 ha drainage (Fredriksen et al. 1975, and Fredriksen and Harr 1979). In very steep, erosion-prone terrain some increase in sediment yield to streams by conventional harvest systems is essentially unavoidable. Studies on watersheds 1, 2 and 3 at the H. J. Andrews Experimental Forest, western Cascade Range, Oregon, exemplify impacts of practices used in the mid-1960's on such terrain. Skyline logging and broadcast burning of the 96 ha Watershed 1 resulted in an approximately 9-fold increase in suspended sediment export in the first 13 years following burning compared with control watershed 2 (Fredriksen & Harr 1979). After 6% of 100 ha Watershed 3 was roaded and 25% clearcut and broadcast burned in three units, suspended sediment export over an 18 year post-treatment period was about 24 times the yield of the control (Fredriksen & Harr 1979). Of the total 12 year post-treatment yield of coarse material from Watershed 3, 99% was exported by two major storms in WY 1965 which triggered numerous debris avalanches (shallow, rapid soil mass movements on hillslopes) and channel scour events.

The overwhelming importance of the two WY 1965 storms in the long records from the H. J. Andrews watershed studies underscores some of the difficulties of interpreting results from the paired watershed study design. Small watersheds are unique in time and space. They have particular physical characteristics and histories of natural and management-related processes.

TABLE 1. SUMMARY OF PACIFIC NORTHWEST STUDIES OF SOIL DISTURBANCE AND COMPACTION BY VARIOUS YARDING SYSTEMS. DATA ARE IN PERCENT OF HARVEST UNIT AREA IN SOIL DISTURBANCE CLASS. SYMBOLS ABOVE COLUMNS DENOTE SOURCE OF DATA. HIGH-LEAD DATA FROM SOURCES * AND ++ ARE AVERAGES OF SAMPLES FROM 3 AND 11 UNITS RESPECTIVELY. TRACTOR DATA FROM ++ AN AVERAGE OF TWO UNITS

Soil Disturbance Class	Yarding System (data source)										
	Tractor-skidder				High Lead		Skyline			Balloon	Heli-copter
	*	**	+	++	*	++	**	+	Δ	ΔΔ	++
Clearcuts											
Undisturbed	36	58			57		73		64	78	
Slightly disturbed	26	14	12	11	22	18	9	8	24	16	2
Deeply disturbed	9	29	16	58	10	11	24	3	5	3	3
Compacted	27	31			9		27		3	2	
Non-soil Area	2				2				4	2	
Selective Cuts											
Undisturbed		43					70				
Slightly disturbed		24					16				
Deeply disturbed		35					13				
Heavy compaction		31					14				

* Dyrness 1965
** F. M. McCorison, pers. commun.
\+ Wooldridge 1960
++ Bockheim et al. 1975
Δ Dyrness 1967a
ΔΔ Dyrness 1972

Losses of N capital from harvesting alone or from harvesting and slash burning removed from 20-132 times as much N as did annual stream runoff in a western hemlock forest in British Columbia (Kimmins & Feller 1976). Grier[3] reported a loss of greater than 95% of N capital in slash by volatilization in fire; on the Entiat Experimental Forest in Washington, 855 kg/ha of N was lost by volatilization from litter and soil. Kimmins and Feller (1976) reported 581 kg/ha of N lost in slash burning and from 92-124 kg/ha of N removed by logging.

Clearcutting alone and clearcutting followed by burning generally result in modest increases in solution losses of N draining below rooting zones or from first-order streams draining small watersheds (Cole & Gessel 1965; Fredriksen 1971; Kimmins & Feller 1976). In one instance, McColl (1978) actually observed a significant decrease in NO_3 following clearcutting. Partial cutting or shelterwood cutting resulted in only a modest increase in N loss in stream runoff (Fredriksen et al. 1975).

In summary, solution losses of N from Pacific Northwest forests which have been harvested generally are small relative to the very large N losses, particularly NO_3^- nitrogen, reported in the eastern US (Bormann et al. 1977). Timber harvesting can accelerate N losses as a result of increased soil removal (Fredriksen 1971). Catastrophic losses of soil from forests can occur both as a result of major storms (Fredriksen 1970) and as a consequence of harvesting or harvesting-related activities such as road building. Soil loss results in a decline of both nutrients and soil productivity until new soil is formed. Consequently, improvements in harvesting techniques which minimize soil loss are desirable.

Nutrient Restitution Following Harvesting

Nitrogen removed from Pacific Northwest forests by harvesting or slash burning can be replaced by N fixation and precipitation or by fertilization. Although fertilizers are widely used in the Northwest and produce a positive growth response, their rapidly rising costs have led to increased consideration of biological N fixation to restore N capital of harvested sites (Bengtson 1979).

Estimates of the quantity of N fixed by trees such as red alder and by shrubs such as snowbrush in Northwestern forests range from 32-325 kg/ha/yr (McNabb, Geist and Youngberg 1977; Newton et al. 1968). The low value reported by McNabb, Geist & Youngberg (1977) is from a snowbrush stand in eastern Oregon where severe moisture stress limits N fixation. Even in that case, N fixation over a 10 year period could result in a stand accretion of 320 kg/ha of N. On a more favorable site in eastern Oregon, Youngberg & Wollum (1976) reported N accretion of 715 kg/ha in a decade.

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[3] C.C. Grier. 1972. Effects of fire on the movement and distribution of elements within a forest ecosystem. Ph.D. thesis. University of Washington, Seattle, Washington.

Scott[4] has shown that snowbrush can increase growth of Douglas-fir seedlings relative to seedlings growing without shade during the first seven to nine years. Red alder is more competitive with Douglas-fir on moist sites. Increasing environmental concern with herbicides is catalyzing research on alternative methods of utilizing the N fixing capacity of red alder and other species by underplanting in already established stands of Douglas-fir. Berg and Doerksen (1975) estimate naturally invading red alder to add from 220 to 848 kg/ha of N to a heavily thinned Douglas-fir stand during a 17 year period. Basal area of the 80-year-old Douglas-fir dominated stand has increased 55% post thinning, not including biomass added by the red alder. In the thinning regime evaluated by Berg & Doerksen (1975), gradual closure of the Douglas-fir canopy is eliminating the alder following the positive benefits of its N accretion.

Longer-term input of N into forests by N fixation also is being studied in older coniferous forests. Nitrogen fixation in decaying coniferous logs occurs at low levels with highest rates in wood with advanced decay (Larsen et al. 1978). Assuming approximately 215 mT/ha of down logs in old-growth Douglas-fir forests as reported by Grier and Logan (1977) and assuming rates reported by Larsen et al. (1978) we estimate 5 kg/ha/yr of N fixation in logs in the western Cascades of Oregon. Trees rooting into rotting logs may utilize this N source. An additional 5 kg/ha/yr of N may be fixed by old-growth canopy-inhabiting lichens (W. C. Denison, pers. commun.). Nitrogen inputs by these long-term processes thus may be several times precipitation and dryfall N inputs and may be important in maintaining productivity of older coniferous forests.

Effects of Fertilization

Fertilization of watersheds has resulted in small but significant increases of inorganic N in streams from both direct fertilizer application into streams and as a result of percolation through the soil mantle (Fredriksen et al. 1975). Cole and Gessel (1965) observed a slight increase in organic N at nearly 1 m depth in the soil mantle following application of 220 kg/ha of urea on one plot and 220 kg/ha of ammonium sulfate on another plot. The maximum loss reported was with ammonium sulfate fertilization, which doubled the total N solution loss. A favorable benefit:cost ratio makes it likely that N fertilization will be widely used in the Pacific Northwest for at least the next decade (Miller & Fight 1979; Bengtson 1979).

NO 400 N/a

Effects on Soil Integrity and Erosion

The sequence of harvest activities affects the physical integrity of soil in two general ways: 1) by in situ alteration of soil physical properties and 2) by acceleration of soil erosion. Each step in the harvest sequence may have distinctive effects on soil physical properties that differ in type, magnitude and duration of impact.

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[4] W. Scott. 1970. Effect of snowbrush on the establishment and growth of Douglas-fir seedlings. MS thesis. Oregon State University, Corvallis, Oregon.

Therefore, in assessing management impact there are certain advantages to process-level studies in which key erosion processes are inventoried on broad time and space scales.

In steep terrain, inventories of debris avalanches over areas of 10 km^2 or more and time periods of a decade or more provide a basis for measuring the impact of roads and clearcuts relative to forested areas on soil erosion by debris avalanches. In six studies in steep land of the Pacific Northwest debris avalanche erosion rate in clearcuts was found to be 0 to 10 times the rate in forested areas and the average increase was 2.4 times (Swanston & Swanson 1976, Ketcheson & Froehlich 1978, F. J. Swanson, unpub. data). Five of these studies include estimates of debris avalanche erosion from forest road right-of-way and rates vary from 25 to 340 and average 125 times the forest area rate (Swanston & Swanson 1976, F. J. Swanson, unpub. data). Of course, these surveys primarily assess effects of some earlier logging and roading methods which have been modified or abandoned, so these figures may not reflect the impact of modern practices. Only continued inventories will measure the benefits of new methods in terms of reduced debris avalanche erosion and associated impacts on streams.

These measures of road and clearcut impacts on debris avalanche erosion reinforce the common belief that roads are the predominant source of erosion accelerated by forest practices. However, although roads have a high impact they affect only a small proportion of the landscape, generally less than 6 or 7% (Froehlich 1978). The smaller, but significant impact of clearcutting influences the remaining 93 or 94% of the landscape. Based on these and several other assumptions, Swanson and Dyrness (1975) estimate that accelerated debris avalanche erosion due to clearcutting alone may roughly equal that of roads.

No studies have contrasted the impact of harvest practices other than clearcutting on erosion by debris avalanches or other single erosion processes. The longer-term soil erosion impacts of timber harvest activities in the Pacific Northwest are unknown, because the natural history of erosion in the region has included periods of accelerated erosion following wildfire. As part of the assessment of long-term impacts, the frequency and erosional consequences of premanagement wildfire should be compared with the frequency and erosional consequences of management activities (Swanson et al., in press). Information necessary to make this comparison is lacking.

Effects on Streams

Small streams draining steep forest land are intimately linked with forest and soil erosion conditions on adjacent slopes. Even in streams as large as third-order the forest canopy may be closed over the stream and erosion from steep hillslopes may be deposited directly into streams. Stream side vegetation affects the stream environment in a variety of ways: it shades streams, thereby regulating water temperature and sunlight available for primary production; supplies nutrients in the form of litter for aquatic invertebrates and microorganisms; stabilizes streambank and bed with root systems; supplies large organic debris to the stream which shapes the physical structure of aquatic habitats and regulates sediment storage in the channel system; retards downslope and downstream movement of soil, sediment,

particulate organic matter and nutrients dissolved in stream and soil water; and forms a distinctive wildlife habitat (Meehan et al. 1977). Upslope vegetation influences streams by regulating runoff characteristics (Harr 1976) and the rate of sediment supply to streams. Consequently, forest practices that affect riparian vegetation and hillslope vegetation and erosion rates have a broad range of impacts on the aquatic environment.

Here we focus only on the effects of timber harvest activities on organic debris loading in streams because this is a dominant management impact on physical characteristics of the stream environment (Swanson et al. 1976; Swanson and Lienkaemper 1978). Effects of forest practices on stream debris vary greatly with timber type, topography and timber felling and yarding systems (Table 2). In studies of ten cable-yarded units in old-growth Douglas-fir in western Oregon Froehlich (1973b) noted negligible changes in coarse (diam.>10 cm) and fine debris loading for units with free-falling buffer strips, but free-felled units without buffer strips experienced about a two-fold increase in coarse debris loading and a five-fold increase in fine organic debris. Cable assist, directionally-felled units without buffer strips had no increase in coarse material but about a three-fold increase in fines.

In spruce-hemlock forests on Prince of Wales Island, southeast Alaska, Swanson & Lienkaemper (unpub. data) observed coarse and fine debris loadings in three high-lead yarded clearcuts that were about 3 and 7 times the levels measured in three uncut streams.

The visual effects of altered organic debris loading in streams are conspicuous, but efforts to determine effects of these alterations on aquatic organisms, sediment routing and development of riparian vegetation are just beginning. In general guidelines for management of stream debris are based on the quantities and size and spatial distributions of debris observed in natural forested streams.

CASE STUDY - H. J. ANDREWS WATERSHED 10

Site Description and Methods

The discussion to this point has been focused primarily on the general effects of harvest and stand regeneration on Northwestern coniferous forest sites. But, generalizations are useful only in that they describe the types of responses one might expect following a harvest disturbance. Differences among sites in topography, soils, climate and vegetation virtually guarantee the land manager a wide range of responses to a given treatment.

In order to give more specific examples of ecosystem response to clearcutting we describe observations from a small watershed in the central western Cascade Range of Oregon which was intensively studied before and after timber harvest. This study exemplified responses to current harvest and regeneration practices.

Watershed 10 is a small (10.2 ha) watershed adjacent to the H. A. Andrews Experimental Forest near Blue River, Oregon. The forest, stream and erosion processes of this watershed were intensively studied before harvest by scientists associated with the Coniferous Forest Biome, US International Biological

TABLE 2. ORGANIC DEBRIS LOADING DATA FOR FOREST AND CLEARCUT STREAM REACHES (STREAM WIDTH 1 TO 8.5 m) IN WESTERN OREGON AND SOUTHEAST ALASKA. BULK DENSITY OF WOOD IN THE OREGON STREAMS WAS ASSUMED TO BE 0.58 AND FOR ALASKA 0.50 g/cm^3

Location	Sample size	Before logging		After logging	
		Coarse*	Fine	Coarse*	Fine
		kg/m²			
Western Oregon+					
Old-growth forest	10	39.1	3.0		
Clearcut					
Free-falling	3	24.9	2.6	56.6	11.5
Cable-assist directional falling	4	50.1	3.8	46.0	12.0
Free-falling, buffer strip	3	38.5	2.5	35.8	3.2
Prince of Wales Island, Alaska∆					
Old-growth forest	3	5.3	1.0		
Clearcut, free-falling	3			15.6	7.1

* Diameter of coarse debris > 10 cm.

+ From Froehlich (1973b).

∆ From Swanson & Lienkaemper (unpub. data).

Program. Research on the forest was focused on its growth, nutrition and water use; stream research was concentrated on sources and utilization of energy and nutrients by stream biota. Of particular concern to both terrestrial and aquatic researchers was the interaction between forest and stream.

Watershed 10 was clearcut in 1975. After clearcutting, both terrestrial and aquatic research continued (first by the CFB and subsequently through a series of grants also funded by the National Science Foundation). The primary objective of post-clearcut research has been to examine the processes involved in re-establishing nutrient cycles with emphasis on vegetation regrowth; soil erosion; and stream biology and chemistry.

Watershed 10 is typical of the steep, deeply incised drainages in this part of the Oregon Cascades. Elevations range from 430 m at the outlet to about 670 m on the upper ridgelines. Average slope of the stream channel is 24°, side slopes range between 25° and 50°.

The climate is typical for the western Cascades. Annual precipitation averages 230 cm of which about 75% falls between October and March, mostly as rain. Mean annual temperature is 8.5°C; mean January and July temperatures are about 0.5°C and 18°C respectively.

Soils generally are deep and well drained. They are classed as typic distrochrepts (Soil Survey Staff 1960) and have an A_1 horizon about 20 cm deep over 50-80 cm of a B1-B2-B3 sequence. Soils are primarily gravelly clay loams with gravel content (>2 mm) ranging from 15-50% of soil volume. The soils are moderately acid; pH averages about 6.0.

Forest floors were classed as duff-mulls (Hoover & Lunt 1952). The average forest floor consisted of an L and F layer about 4-5 cm thick. No H layer was present. Numerous fallen logs were present on the soil surface.

Before logging the watershed was dominated by a 450-year-old site class II-III (Dilworth 1974) Douglas-fir forest. Forest environments within the watershed ranged from relatively hot and dry to cool and moist (Grier & Logan 1977). These environmental differences were reflected mainly in the composition of understory vegetation. In drier habitats the understory was dominated by rhododendron, salal and beargrass (*Xerophyllum tenax*). In the more mesic habitats, the understory was dominated by rhododendron, salal and/or Oregon grape. Understory vegetation in cool-moist environments was mainly swordfern and a variety of herbs such as twinflower (*Linnaea borealis*). A well developed understory tree stratum was present in all environments.

Table 3 shows the distribution of organic matter and N in various components of vegetation on Watershed 10 prior to and one year after harvest. Accumulations of living biomass and N contained in this biomass were large, especially in comparison with younger Douglas-fir stands (Cole et al. 1967, Turner[5]). Accumulations of organic matter and nutrients in detritus on the

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[5] J. Turner. 1975. Nutrient cycling in a Douglas-fir ecosystem with respect to age and nutrient status. Ph.D. thesis. University of Washington, Seattle, Washington.

TABLE 3. DISTRIBUTION OF ORGANIC MATTER AND NITROGEN IN VARIOUS COMPONENTS OF A 450-YEAR-OLD DOUGLAS-FIR FOREST ON WS 10, H. J. ANDREWS EXPERIMENTAL FOREST BEFORE AND ONE YEAR AFTER CLEARCUTTING

Component	450-year-old Forest		One Year Clearcut	
	Organic Matter*	N**	Organic Matter	N
	kg/ha			
Trees				
Foliage	12,400	128	0	0
Woody	699,000	391	0	0
Understory				
Foliage	1,470	15	359+	4+
Woody	5,310	3	850+	1+
Roots	152,000++	198++	146,000++	205++
Total Vegetation	870,180	735	147,209	210
Detritus				
Forest Floor	51,200	256	30,870ΔΔ	171ΔΔ
Logs, snags, stumps	215,000	215	103,000ΔΔ	51ΔΔ
Soil	133,000Δ	3,724Δ	156,077ΔΔ	3,902ΔΔ
System Total	1,269,380	4,930	437,156	4,334

* From Grier & Logan (1977)

** C. C. Grier, unpub. data; methods outlined by Grier (1977)

\+ From Gholz, Hawk & Campbell (1977)

++ From Santantonio, Hermann & Overton (1977); with no decomposition of large roots during the first year and 25% annual decomposition for fine roots (Sollins, Cromack, Fogel & Li, in press).

Δ From Sollins et al. (in press)

ΔΔ K. Cromack, Jr., unpub. data

soil surface were also large compared with younger forests (Harris et al. 1973; Cole et al. 1967). Forest floor material and standing and fallen dead trees amounted to 21% of the total soil organic matter accumulation, and 9.5% of N accumulation.

Soil organic matter constituted only 10.5% of the total organic accumulations in the ecosystem prior to clearcutting. In contrast, the major N reserve was the soil, which contained 75.5% of the total system N.

Leaf biomass and leaf area (all sides of all leaves) on the watershed were comparable with values reported for other mature Douglas-fir stands (Grier & Running 1977; Gholz et al. 1976). In contrast with many younger forests, 12% of leaf biomass and about 15% of total leaf area were in the understory of this old-growth forest.

Aboveground net primary production on the watershed averaged 7970 kg/ha/yr while belowground net production was about 2900 kg/ha/yr. Net production was entirely detritus; living biomass declined at the rate of about 8400 kg/ha/yr during the study. This is in distinct contrast with younger stands in which 70-85% of net production accumulates annually as woody biomass (Grier & Logan 1977).

In spite of low production, decreasing biomass and the large amounts of decomposing detritus, the ecosystem was basically conservative of nutrients. Inputs of N in precipitation and dust averaged 2.0 kg/ha/yr; N-fixation by canopy-inhabiting lichens added an additional 2.7 kg/ha/yr (Sollins et al. in press).

Overall, the old-growth forest of Watershed 10 had several unique attributes and several characteristics similar to those of younger forests. Unique aspects were the large accumulations of biomass and detritus and a production economy dominated by detritus. The old-growth forest was similar to younger ones in conserving essential nutrients.

Harvest and Site Preparation

During the summer of 1975, Watershed 10 was clearcut. No roads were constructed on the watershed itself but two landings were built on the ridgeline. Trees were felled upslope and then yarded using a skyline cable system. Cull logs and a large proportion of the fallen logs from earlier mortality also were yarded to the landing and sorted. Unusable material was piled near the landings for later burning; merchantable material was placed on trucks.

Slash remaining after "clean yarding" the watershed was not burned. The following spring the cutover area was planted with 2-0 bareroot Douglas-fir seedlings on a 2.4 x 2.4 m spacing.

Harvest - Removal of Vegetation, Debris and Nutrients

Yarding of material from the watershed resulted in the removal of 815 mT/ha of organic matter, or 64.2% of the total organic matter in the

system (Table 3). However, only 574 kg/ha of N were removed by harvesting, or about 12% of the total N in the system. Residual aboveground live biomass on the watershed one year after logging totaled only 0.16% of prelogging vegetation biomass. Roots constituted the largest single organic matter pool following logging. Soil organic matter was assumed to increase by approximately 23,077 kg/ha or 17% due to stirring of soil and mixing in of forest floor by logging.

During the first year after logging, uptake of N by residual vegetation is estimated to have been approximately 4 kg/ha, assuming essentially complete new foliage. Immobilization or even temporary uptake of N may have occurred in woody residues, both above and belowground. Sollins, Cromack, Fogel and Li (in press), found decomposing fine roots of Douglas-fir to increase 19% in total N mass during first-year decomposition, indicating that rapid fine root decay may be a temporary "sink" for soil N. Coarse roots take much longer to decompose, hence, would be a long-term input to soil organic matter (S. Cline, personal commun.).

Loss of N dissolved in stream runoff totaled 0.8 kg/ha the first year post-clearcutting (P. Sollins, personal commun.). Export of particulate organic N totaled 2.0 kg/ha during the first year post-clearcutting (Swanson, personal commun.).

Erosion activity in Watershed 10 has been monitored since cutting in order to assess the effect of deforestation and ecosystem recovery on each erosion process and total sediment yield. With only three years of post-cutting data available, it is premature to assess management impacts in detail. However, preliminary evaluation of bedload export for WY 1976 through WY 1978 does yield interesting insights and hypotheses.

Four storms during WY 1976 washed 17 T of particulate into the sediment basin at the base of the watershed. The first two storms produced peak flows that typically occur several times a year, yet they exported 6.1 T of bedload which is nearly 7 times annual export for old-growth forest conditions (Swanson et al. in press). About two-thirds of this material was organic matter, mostly needles and twigs input to the stream during logging operations. The third storm was larger and it exported 7.0 T of bedload of which 31% was organic matter. The fourth storm produced a still larger peak flow but this event exported only 3.5 T of bedload material including only 17% organic matter. Apparently by this time, much of the readily transported organic material had been flushed downstream to the sediment basin or it was deposited in more stable debris accumulations within the channel. Total bedload export during the first year following clearcutting was 19 times greater than the estimated average annual value for the forested condition (Swanson et al., in press).

Water Year 1977 was the driest in the 86 year history of precipitation records in central western Oregon, and there was no significant bedload discharge from the watershed. During WY 1978 several major events caused deposition of 8.9 T of material in the basin.

These results follow two general patterns: 1) a decline in total export through the sequence of major storms and 2) a decrease during a season in the proportion of organic matter in major export pulses. Water Year 1976 experienced about twice the total export as WY 1978, although the later period included the two highest flows since clearcutting occurred.

The impact of logging practices on particulate matter export can be considered in terms of two factors: 1) availability of material to be moved and 2) occurrence and magnitude of storm flow events necessary to move it. Timber harvest activities may affect both sediment availability and watershed hydrology. In several of the experimental watershed studies in western Oregon, forest practices, in particular construction of an extensive road system, have resulted in increased peak flows for some types of events (Harr 1976). However, for Watershed 10 following clearcutting Harr & McCorison (1979) have noted decreased peak flows for events involving snowmelt and no detectable changes in peak flow for rainfall only events. Therefore, changes in sediment export following clearcutting primarily reflect changes in storage and availability of materal for export rather than altered watershed hydrology.

Based on study of organic debris storage in the channel and channel geometry, export from Watershed 10 apparently will come from three sources, each associated with a specific time frame: 1) material input to the channel during the logging operation itself which was mainly fine, green organic matter (material larger than about 5 cm diameter, 50 cm length was hand cleaned from the channel) and some mineral soil; 2) material which had entered the channel by natural processes before logging and had been in temporary storage behind obstructions (logs and root wads) in the channel, but was released from storage when the obstructions were yarded out of the channel; and 3) material input to the channel by hillslope erosion processes following logging. There may be a general phasing of export of these three sources of material with source 1 mainly contributing material to watershed export in the first one to three years following cutting; source 2 gaining importance in the latter part of this period; and post-logging hillslope erosion (source 3) becoming the dominant source several years after cutting. This routing of material through a watershed is an important element of ecosystem response to disturbance and it is relevant to interpretation of sediment yield data from experimental watersheds. For example, sediment yield is commonly interpreted in terms of hillslope erosion where in many cases initial post-disturbance pulses of sediment primarily reflect changes in channel storage. These interpretations must be based on a whole-system level understanding of soil and sediment routing as depicted in Fig. 2.

Harvest Disturbance and Compaction of Soil

Following yarding of Watershed 10, soil disturbance and compaction were inventoried using a continuous line transect sampling method (Harr & McCorison 1979). In Watershed 10 following logging 70% of the area was uncompacted, 10% slightly compacted, and 20% deeply compacted, based on evidence of more than 3 passes of equipment or logs. Undisturbed areas covered 50% of the watershed, 16% was lightly disturbed, and 34% deeply disturbed. These figures indicate a much greater degree of soil compaction and

disturbance than that reported from previous studies of skyline yarding for clearcuts and thinning operations (Table 1). Both heavily compacted and deeply disturbed soils on Watershed 10 were about 7 times more extensive than in the skyline clearcut studied by Dyrness (1967a). The definition of terms in these two studies was essentially the same, so these marked differences must derive from real contrasts in conditions or differences in field interpretation.

The topography and landing location in Watershed 10 did not permit full suspension of logs over all areas of the watershed. Topography and landing location resulted in many logs being dragged across the southeast slope, exposing mineral soil and in some instances uprooting stumps. Logs on the northwest side were yarded away from the slope, so ground disturbance was minimal.

Effects on Streams

We determined changes in organic debris loading in and adjacent to the stream on Watershed 10. Before logging coarse (diameter > 10 cm) debris in a 3 m wide band along the stream was 8.8 m^3/100 m of channel length in the main branch and 18.5 m^3/100 m in tributary channels (Froehlich et al. 1972). After logging the main branch contained 17.2 m^3/100 m and the tributaries had 25.3 m^3/100 m (H. A. Froehlich, personal commun.).

Coarse debris loading in the stream corridor increased by about 30 to 95%, in spite of efforts to keep debris from entering the channel by directional felling with hydraulic jacks. Side slopes of up to 70% and the heavy down hill lean of numerous, massive old-growth trees contributed to increased debris loading in the stream area. In contrast, the terrestrial standing crop of down logs was reduced to 22% of the pre-logging level (Table 3).

Most of the recorded increase in debris loading is material on the stream bank rather than in the active channel. The annually active streambed experienced a reduction of coarse debris loading. About half of the large debris pieces which trapped sediment in the main channel before logging were removed for utilization or fire hazard reduction. Hand cleaning of the channel involved placing slash in piles along the stream bank. These practices resulted in heavy slash concentrations (> 30 cm deep) bordering 50% of the stream perimeter. The deeper slash accumulations along the channel margins are trapping sediment transported downslope by surface erosion processes. These slash deposits, composed primarily of piles of limbs, also affect development of riparian vegetation by forming sites unsuitable for establishment of many important riparian species, such as red alder, which commonly establish on bare mineral soil or deposits of sediment. The long-term effects of these conditions on sediment routing, riparian vegetation development and aquatic organisms are being followed as part of the study on response of the Watershed 10 ecosystem to clearcutting.

Summary of Harvest Effects on Watershed 10

The major removal of N from the watershed was the export of logs. Watershed output of N by stream transport of soluble and particulate transport

has been very modest, totalling 2.8 kg/ha during the first year. All losses of N from harvesting, solution and erosion were 577 kg/ha, which represented 11.6% of the total system N capital.

Sediment losses increased 19 times over the pre-cutting annual average for the forested condition (Swanson et al. in press). Increased erosion resulted from both removal of pre-logging debris dams and increased sediment deposition within the channel.

Harvest effects on soils in Watershed 10 resulted in 20% severe compaction, less than half that of typical tractor-skidder logging. Heavy soil compaction was about four times more extensive on Watershed 10 than in the skyline clearcut studied by Dyrness (1967a); the topography and landing location in Watershed 10 did not permit full suspension of logs over all areas of the watershed.

Debris loading along the streambanks increased 130 to 195% over pre-logging conditions, while coarse debris loading of the stream decreased. The deeper slash deposits bordering 50% of the stream margin are trapping sediment transported downslope by surface erosion processes, while inhibiting establishment of riparian vegetation such as red alder.

ALTERNATIVES TO CURRENT PRACTICES

Future Management Considerations for Nitrogen Cycling

Typical harvest and slash burn practices in Pacific Northwest forests remove less than 20% of the total system N capital. Provided soil compaction and erosion are minimized by use of logging methods appropriate to the terrain, maintenance of sustained productivity of forested lands depends upon restitution of essential nutrients in sufficient quantities. Nitrogen capital can be restored either by fertilizer or by N fixation; however, increased energy costs and priority use of fertilizers for food production (Bengtson 1978, 1979), may encourage renewed silvicultural interest in N fixation. Future questions which need research include how to manage the various kinds of N fixers available in the Pacific Northwest, including red alder, snowbrush and both native and introduced legumes, to best realize the potential benefits of such species. If the goal of forest management is maintenance of sustained site productivity, then different time allowances for managing secondary succession, including N fixation, may be necessary for different site qualities. Integrated studies of conifer seedling survival in the context of both nutrient availability and potential competition are needed if proper evaluation of the benefits: cost of N fixation are to be adequately assessed. Modern methods for studying plant-water relations and C allocation should be utilized in assessing the reality of competition by N fixing species with conifers. Furthermore, animal damage studies should consider the possible protective benefits of brush species such as snowbrush, which usually are considered strictly as competing species. The potential silvicultural benefits of brush in protecting sites from excess erosion while desirable conifer stands are being re-established should also be considered (Youngberg 1966).

Management of Soils

Management strategies designed to minimize soil losses benefit both forest and aquatic resources by maintaining water quality and productivity of tree-growing sites and reducing disturbance of the stream environment. Forest land managers in the Pacific Northwest are currently experimenting with methods to decrease erosion accelerated by roads and timber harvest activities. Improved road placement, design, construction methods, and maintenance have been a big part of this effort. Common techniques involve keeping roads high on ridges, minimizing construction of fills by "end-hauling" and "fully benching", and maintaining road drainage systems, particularly during large storms.

Improvements in yarding system capabilities, such as longspan skyline, reduce road mileage and the amount of soil damage due to the logging system itself. In the very steep, debris avalanche-prone lands in portions of the Oregon Coast Range, forest land managers are experimenting with felling, yarding and slash disposal systems designed to reduce the occurrence of soil mass movements from the heads of small streams. Where wind throw potential of residual trees is judged to be low, small stands of trees are left in these critical headwall areas. Other headwalls are cut with directional felling techniques, but they are not broadcast burned in order to maintain the soil binding root systems of residual brushy vegetation.

In spite of expanded use of these methods, clear assessment of their benefits has not been conducted. Inventory of erosion under forested conditions and following various roading and logging practices should be carried out to determine which erosion processes are dominant and what are the erosional and economic benefits of innovative harvest and roading systems employed to minimize erosion. Such inventory systems are being developed by several National Forests in the Pacific Northwest and Rocky Mountains. In these areas of steep terrain, inventories of debris avalanches, based on studies of aerial photographs and updated by field surveys following major, debris avalanche-triggering storms, provide a basis for estimating erosion from different types of roads and harvest systems.

On a broader scale, forest land managers should plan soils management on the scale of large drainage basins, since management activities affect soil and sediment routing on this scale. At present, these effects are randomly distributed and efforts to minimize these impacts are based on localized, site specific decisions. Perhaps in the future we will have the capability and inclination to manage soil and sediment routing on a broader scale. In this view, watersheds could be considered a mosaic of roaded, forested and regenerating sites each with a characteristic susceptibility to erosion determined by natural and man-induced factors. Management of such a landscape would have the objective of distributing zones of high erosion potential geographically and over time, in order to minimize concentration of land in highly erodable conditions near sensitive stream reaches.

Future Management Considerations for Streams

Over the past decade federal and state legislation have encouraged the protection of small streams which make up such a large proportion of the total length of river systems. In steep forest land small streams are particularly sensitive to changes in vegetation and soil conditions on adjacent hillslopes. Therefore, management of streams involves not only consideration of in-stream factors, but also variation in long-term vegetation and soil erosion conditions on hillslopes.

A key element of small streams in the forest environment is woody organic debris. Management activities may greatly increase or decrease total organic debris loading and alter the size distribution of in-stream organic debris. Currently suggested guidelines for stream debris management in the Pacific Northwest are to leave the natural pieces of debris which had entered the channel before logging, to introduce no new organic material during logging operations, and to plan for future large debris inputs to streams (Swanson & Lienkaemper 1978). In very steep terrain, however, these goals are difficult to accomplish, as the Watershed 10 example demonstrates.

Stream protection may be provided by buffer strips of vegetation along streams. Although buffer strips usually are justified on the basis of minimizing stream water temperature increases, streamside vegetation has a great variety of effects on the aquatic ecosystem (Meehan et al. 1977). Shading by streamside vegetation reduces sunlight available to drive in-stream primary productivity. Riparian vegetation provides coarse and fine litter for the stream that serves as a primary source of nutrients for aquatic invertebrates and shapes the physical structure of the stream ecosystem. This vegetation also stabilizes banks and retards downstream and downslope movement of soil and sediment. Consequently, in judging the value of buffer strips their full range of functions for the stream ecosystem should be considered. Some of these functions may continue to be carried out even if a buffer strip experiences blow down.

Recognition of the role of streamside vegetation as the source of large organic debris for streams raises some questions concerning long-term implications of forest practices (Swanson & Lienkaemper 1978). Good silvicultural practices are designed to minimize production of unutilized woody debris by frequent thinning and harvesting. Unless streamside vegetation is specifically managed with one goal being production of large woody debris for the stream, levels of debris loading in streams will gradually decline as natural debris and slash from the first logging entry is decomposed and transported downstream. Reduced debris loading will result in altered aquatic habitat, reduced in-stream sediment storage associated with large organic debris, and increased rate of sediment movement through stream systems. The biological and physical consequences of these broad scale changes in stream systems are unknown.

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Original

Managing Soil Moisture to Control Soil Compaction

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Managing soil moisture to control soil compaction represents an opportunity for foresters and loggers alike to harvest timber with skidding vehicles with less soil compaction. The principles upon which managing soil moisture is based are simple and straightforward. Rather than being a constraint on a logging operation, managing soil moisture offers greater opportunity to use skidding vehicles without normally increasing the amount of soil compaction. To manage soil moisture effectively during timber harvesting, an organization has to be somewhat flexible and opportunistic. Thus, industry, conducting their own operations, has a greater potential to manage soil moisture than an organization managing through contracts. Whatever the situation, advocating the management of soil moisture during harvesting can only provide additional protection for the land resource.

Managing soil moisture to control soil compaction is developed from the concepts of soil compaction presented in, "Compaction of Southwest Oregon Soils." Skidding vehicles applying pressure to undisturbed forest soils will increase the density and shear strength of the soil. Shear strength will increase until it can support the pressure applied. The amount of compaction occurring is primarily a function of the pressure applied and to a lesser extent, the soil moisture content. Soil moisture content apparently is not a factor in the compaction of some soils.

While the importance of moisture content as a factor of soil compaction by skidding vehicles is being de-emphasized, managing soil moisture is still very important for protecting soils from puddling. Soil puddling is perceived as being far more damaging to soil productivity than compaction alone. Puddling is a serious problem because: soil aggregates are destroyed; natural failure surfaces between soil particles are damaged, resulting in dry soil forming clods; soils lose their ability to support skidding vehicles resulting in the formation of ruts and damage to residual root systems; soil aeration is impaired; and recovery of soil to natural density can take longer because the failure planes between aggregates have been destroyed.

A distinct boundary does not exist between when is a soil being puddled versus compacted. Aggregate shear strength decreases before maximum density of the soil is reached at low compaction efforts (Fig. 1). The moisture content at which aggregates lose strength depends on the soil. The greatest loss of aggregate shear strength for medium and coarse textured soils apparently occurs around field capacity. While no data is available, aggregate strength of fine textured soils, particularly those with expandable clay minerals, would decline over a wider range of moisture tensions. The loss of aggregate shear strength would probably begin at moisture tensions below field capacity. The measurement of moisture tensions seems to be a more logical way of testing soils for puddling than moisture content measurement but the concept will have to be field tested before it becomes practice.

The objective for managing soil moisture is to reduce the risk of puddling soil and to reduce the moisture content of moisture sensitive soils. The key elements of managing soil moisture are: principles of soil moisture management, scheduling harvesting operation, spring logging, fall logging, and site preparation.

Principles of Soil Moisture Management

The potential for minimizing soil compaction by managing soil moisture is extremely good in the Pacific Northwest, particularly in Southwest Oregon. The long, dry summers with only limited precipitation requires the forest to use water stored in the soil. Once used, there is little opportunity for the water to be replaced. At low elevations, potential evapotranspiration normally begins to exceed precipitation in May (Fig. 2). Soil moisture drawdown begins early on these sites. At mid-elevations, potential evapotranspiration may not exceed precipitation until June (Fig. 3) and at higher elevations, not until July (Fig. 4). Thus, Southwest Oregon offers considerable opportunity for using skidding vehicles on relatively dry soils.

Vegetation, primarily the trees in forest environments, is the wick removing water from soil. Felling trees severs the wick and drastically reduces the rate of water loss from a soil. Therefore, planning the harvest of timber stands so that they may remove as much water from the soil as possible before they are fell is the key to protecting soils from puddling and reduce compaction of moisture sensitive soils.

To illustrate the importance of trees for reducing soil moisture (Fig. 5), data from the H. J. Andrews Experimental Forest shows soil moisture in the top one foot of soil to be much higher in a clearcut the first season after logging as compared to the soil moisture present in an adjacent old-growth forest. In this example, soil moisture decreases to field capacity in early June for both the forest and clearcut; however, a few precipitation events occur in June and July which keep the moisture content of the clearcut soil at or above field capacity until mid-August. In contrast, transpiration from the trees keep soil moisture storage in the forest below field capacity from early June until late October when enough rain finally falls to recharge the profile. During the same period, moisture storage in the clearcut is below field capacity for only a little over a month.

Some soil moisture drawdown can be expected to occur on a logged site by evaporation or transpiration by remaining vegetation. Southerly aspects are more apt to dry than northerly aspects. In the example, the clearcut site had already been logged and broadcast burned; thus, the soil had minimum cover. Prior to logging, logs and slash can insulate the soil and keep soil evaporation low. Therefore, soil moisture should remain very near pre-logging moisture contents once the trees are fell unless the transpiration potential of remaining vegetation is high.

Once the timber is fell, precipitation can recharge the soil and soil moisture can be expected to remain high thereafter. More precipitation is retained by the soil, litter and logging slash of felled timber than a standing forest canopy because a forest does more than transpire water to lower soil moisture. The canopy of standing trees intercepts a portion of each precipitation event and returns it to the atmosphere, preventing it from ever reaching the ground. Under a forest canopy, the litter can function as another evaporative surface. These benefits of maintaining a forest canopy is evident in the data by the rapid fluctuations in the clearcut soil moisture data whereas soil moisture storage under the forest changes more slowly.

Scheduling Harvesting Operations

To take advantage of transpiration by trees reducing soil moisture, felling should be delayed until just prior to skidding. Delayed felling will allow for the greatest reduction in soil moisture and reduce the risks of precipitation occurring on felled timber before it can be skidded.

(those devel on ash, pumice (non cohesive) and those w/2:1 clays that are cohesive)

Moisture insensitive soils can be skidded earlier in the summer than moisture sensitive soils without causing as much compaction. In planning a summer's work schedule, sites with moisture insensitive soils should be scheduled before sites with moisture sensitive soils at similar elevations. The felling of sites with moisture sensitive soils should be delayed until late summer, allowing them the longest time to dry; however, logging of moisture sensitive soils should not be delayed until fall when the probability of fall precipitation increases and the risk of compacting moist soils is again high.

In some areas, a summer's logging schedule may not include both moisture sensitive and insensitive soils. If silvicultural prescriptions and soils are similar, scheduling units on southerly exposures before those on northerly exposures may have some slight advantages for reducing soil compaction. The scheduling of units with different silvicultural prescriptions also offers an opportunity to reduce the risk of logging on soils which may puddle or compact. Partial cut units should be entered first because the remaining overstory can continue to transpire water and reduce soil moisture. A relative ranking of silvicultural prescriptions and transpiring potential of the remaining vegetation for minimizing soil puddling and compaction follows:

Order	Stand Prescription and Condition
1.	All cable yarded units - soil compaction will be least important.
2.	Light partial cuts with a significant amount of brush in the understory.
3.	Light partial cuts without brush and heavier partial cuts with brush.
4.	Shelterwood units (8-20 trees per acre) with brush.
5.	Heavy partial cuts.
6.	Clearcuts with heavy brush.
7.	Shelterwood units without brush.
8.	Clearcut units without brush.

Clearcut harvest units represent the greatest risk for soil puddling from subsequent precipitation after the trees are fell. Therefore, the best time to harvest clearcut units is middle to late summer when the probability of precipitation is lowest. Clearcut units should be skidded soon after they are fell while the soils are dry.

Spring Logging

The major problem with spring logging is that it begins on a soil which has been fully recharged during the winter and the probability of recieving more precipitation is still high. To help minimize these two factors, trees should not be fell early; felling should be coordinated to coincide closely with skidding. Early season logging should be done on moisture insensitive soils first and partial cut units should be logged before first entry shelterwood or clearcut units.

Coordinating felling and skidding operation with only a short time lapse between is one of the more important management practies which can help minimize soil puddling and compaction associated with spring logging.

Fall Logging

Fall logging represents the very best and some of the worst conditions for skidding vehicles to compact or puddle soil. Prior to fall precipitation, the soils are at their driest and compaction is at a seasonal low. But the probability of precipitation returning soil moisture to the wet end of the scale increases daily. Therefore, the dilemma is what to do when it rains.

When it rains, the most important question to ask is how much. Foliage and litter intercept precipitation and reduce the amount reaching the soil. Deep litter may intercept most of the first fall storms and the soils may remain nearly dry. Having a raingauge on or near a logging site would be usefull for making a quick assessment of the situation.

If precipitation has reached mineral soil, the depth to which the wetting front has penetrated the soil can be measured with a shovel. Moisure penetration will be greatestin skid trails, landings, and areas without a cover of litter and slash. Advancement of the wetting front is also highly variable and should be checked in several places to get an average.

What happens if recent precipitation has entered the soil 2- to 4 inches? The first thought may be to close down and operation. But the soil below the wetting front is still dry and will compact little different than before the rain. If it is early in the fall, the probability of more precipitation is less than later. Therefore, some benefit may be gained by stopping logging for a couple of days early in the fall and allow the soil to drain. If the probability of more precipitation is high, a delay may only make the situation worse.

Shallow penetration of precipitation over dry soil may be viewed another way. A delay may make the situation worse, so how serious is the puddling and compaction to the soil if logging to finish a unit is allowed to continue. The puddling may look bad, but it is confined to only the surface few inches. A portion of the surface soil is always displaced so that part of the puddled soil will probably be removed this way. The surface is where freezing and thawing, and wetting and drying cycles are most active. A thin layer of puddled soil would partially recover fairly quickly; deeper conpaction and puddling would be a more serious,long term problem. The depth to which recovery processes are most active could be used as a guide for suggesting how deep moisture would be allowed to penetrate before it is considered serious.

Site Preparation

Mechanical site preparation and fuels management represents a greater potential for soil compaction and puddling than does timber harvesting alone. Machines used for site prepatation will generally cover more of an area than does skidding vehicles and their tracks are more apt to be in direct contact with mineral soil since the slash and some of the litter is being pushed ahead of the machine.

Mechanical site preparation should be done the same season as the logging. Unless considerable overstory and understory vegetation remains on the site after logging, the soils will recharge following winter precipitation and be extremely slow to dry. Thus, site prepatation can cause considerable soil compaction and puddling if attempted the season after logging.

If mechanical site preparation is not completed before fall precipitation begins, some other form of site preparation, such as prescribed fire ought to be used on units where most or all of the overstory has been removed and little understory remains.

Summary

The planning of felling schedules and coordination of felling and skidding operations to maintain the transiration capacity of trees as long as possible before logging represent an opportunity that foresters and loggers have available to manage soil moisture. The result of managing soil moisture will be less puddled soil and less soil compaction, particularly on moisture sensitive soils, while doing the same work. Some side benefits may be a little longer logging season and more efficient machine operation because of less track and wheel slip. The cost of a program to manage soil moisture is neligible. All it requires is a little organization planning each spring as to the order units will be harvested and coordinating the operation of felling and skidding crews.

Managing soil moisture will not always be successfull. Over the short-run, even the best plans will be rained out because the concept is based on the probability of precipitation occurring. Over the long-run, the probability of having a long, dry summer is very good and therefore, a plan to manage soil moisture to minimize soil compaction will be successfull.

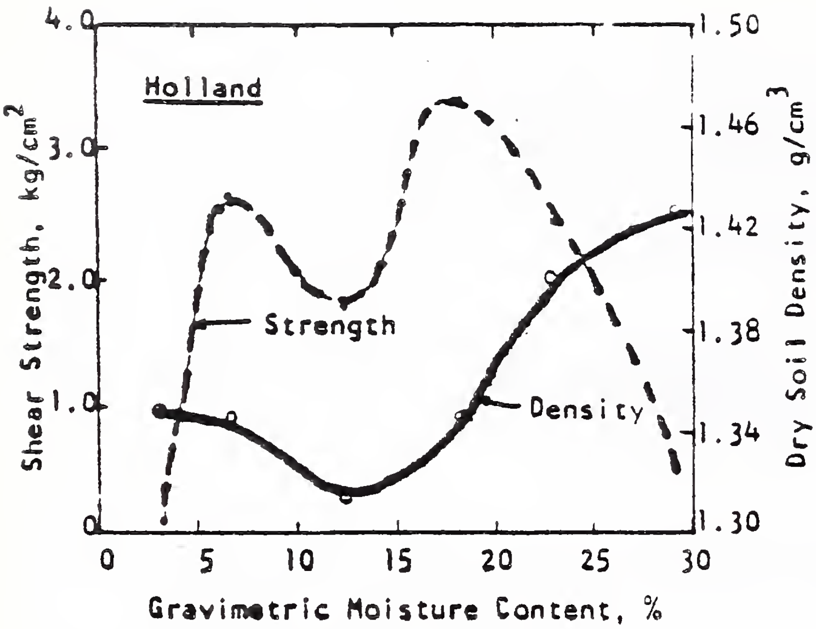

Fig. 1. Shear Strength and Dry Soil Density Changes with Moisture Content of the Holland Soil at Low Compactive Effort (5- blow count).

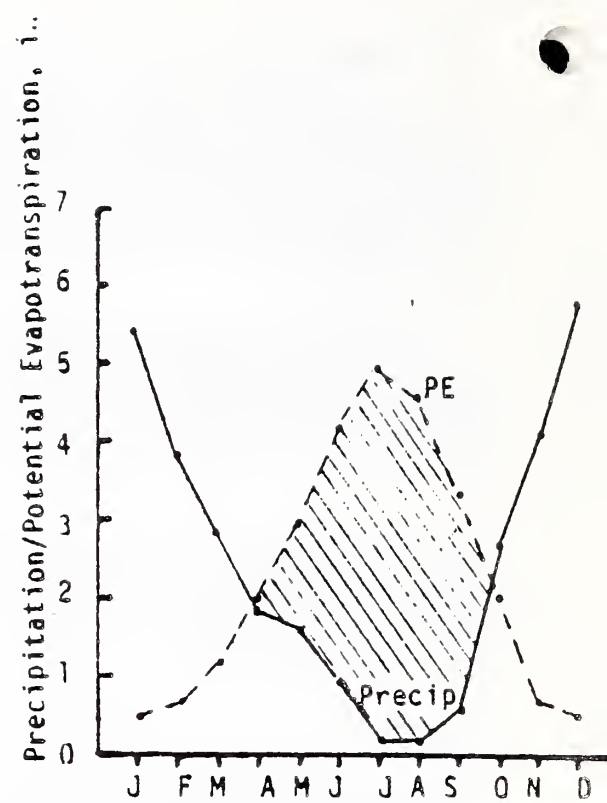

Fig. 2. Monthly precipitation and potential evapotranspiration at a low elevation site (Grants Pass, 950 ft.).

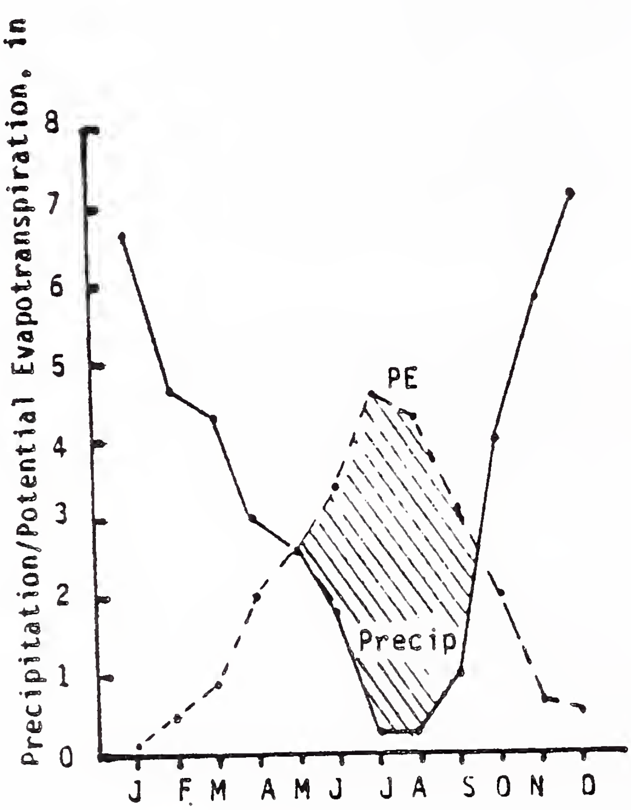

Fig. 3. Monthly precipitation and potential evapotranspiration at a middle elevation site (Prospect, 2500 ft.).

USDA FOREST SERVICE GENERAL TECHNICAL REPORT PNW-24

ENVIRONMENTAL EFFECTS OF FOREST RESIDUES MANAGEMENT IN THE PACIFIC NORTHWEST

a state-of-knowledge compendium

PACIFIC NORTHWEST
FOREST AND RANGE EXPERIMENT STATION
U.S. DEPARTMENT OF AGRICULTURE
FOREST SERVICE PORTLAND, OREGON

Forest Sci., Vol. 25, No. 3, 1979, pp. 427–439

Broadcast Burning: 25-Year Effects on Forest Soils in the Western Flanks of the Cascade Mountains

JAMES F. KRAEMER
RICHARD K. HERMANN

ABSTRACT. To determine the long-term effects of broadcast burning on the physical and chemical properties of forest soils, soil samples were collected from paired burned and unburned plots established between 1947 and 1953 after clearcutting in the western Cascade Mountains of Oregon and Washington. Soils were analyzed for organic matter, total nitrogen, phosphorus, potassium, and calcium, as well as permeability and wettability. Results failed to show statistically significant differences between properties of burned and unburned soils, suggesting that broadcast burning does not have a lasting effect on chemical and physical properties of soil. FOREST SCI. 25:427–439.

ADDITIONAL KEY WORDS. Chemical properties, nutrient status, permeability, wettability, fire.

WILDFIRE has been a natural element in the development of many stands in the western Cascade Mountains of Oregon and Washington. Since the advent of forestry practices in the Pacific Northwest, much effort has been directed towards suppressing wildfires. Conversely, forest managers have used prescribed fire after logging for slash disposal and site preparation. The widespread use of prescribed burning has warranted study of its possible impacts on soil nutrients and physical properties. Numerous publications discuss the extremely varied effects of controlled burning and wildfires on forest soils (Kozlowski and Ahlgren 1974). Partly at least, this variation may be explained by the mosaic of conditions, from unburned to severely burned, caused by a fire. Some of the nutrient capital in logging debris, the litter layer, and soil unquestionably volatilizes during controlled burning and later leaches from the ashes.

The loss of organic matter in the soil may be one of the most significant consequences of fire because organic matter plays an extremely important role in maintaining the fertility and favorable physical condition of soils. Slash burning in the Douglas-fir region of Washington (Isaac and Hopkins 1937, Youngberg 1953) and Oregon (Austin and Baisinger 1955, Dyrness and Youngberg 1957) reduced organic matter as much as 75 percent, depending on the severity and soil depth of the burn. The Oregon studies also indicated that organic matter in burned soils builds up slowly—2 years after burning, organic matter in the burned soil was still less than half that in the unburned soil.

Because organic matter contains most of the nitrogen in soils, burning affects both similarly. Severe burning that destroys large amounts of organic matter

The authors are Graduate Research Assistant and Professor of Forest Ecology, School of Forestry, Oregon State University, Corvallis, Oreg. 97331. This is Paper 1156 of the Forest Research Laboratory, Oregon State University. The authors wish to thank the Pacific Northwest Forest and Range Experiment Station of the U.S. Forest Service for the data, photographs, and assistance, as well as funds through a co-op aid agreement with Oregon State University. Special thanks are extended to Owen Cramer for his cooperation and helpful guidance. Manuscript received 21 July 1978.

United States
Department of
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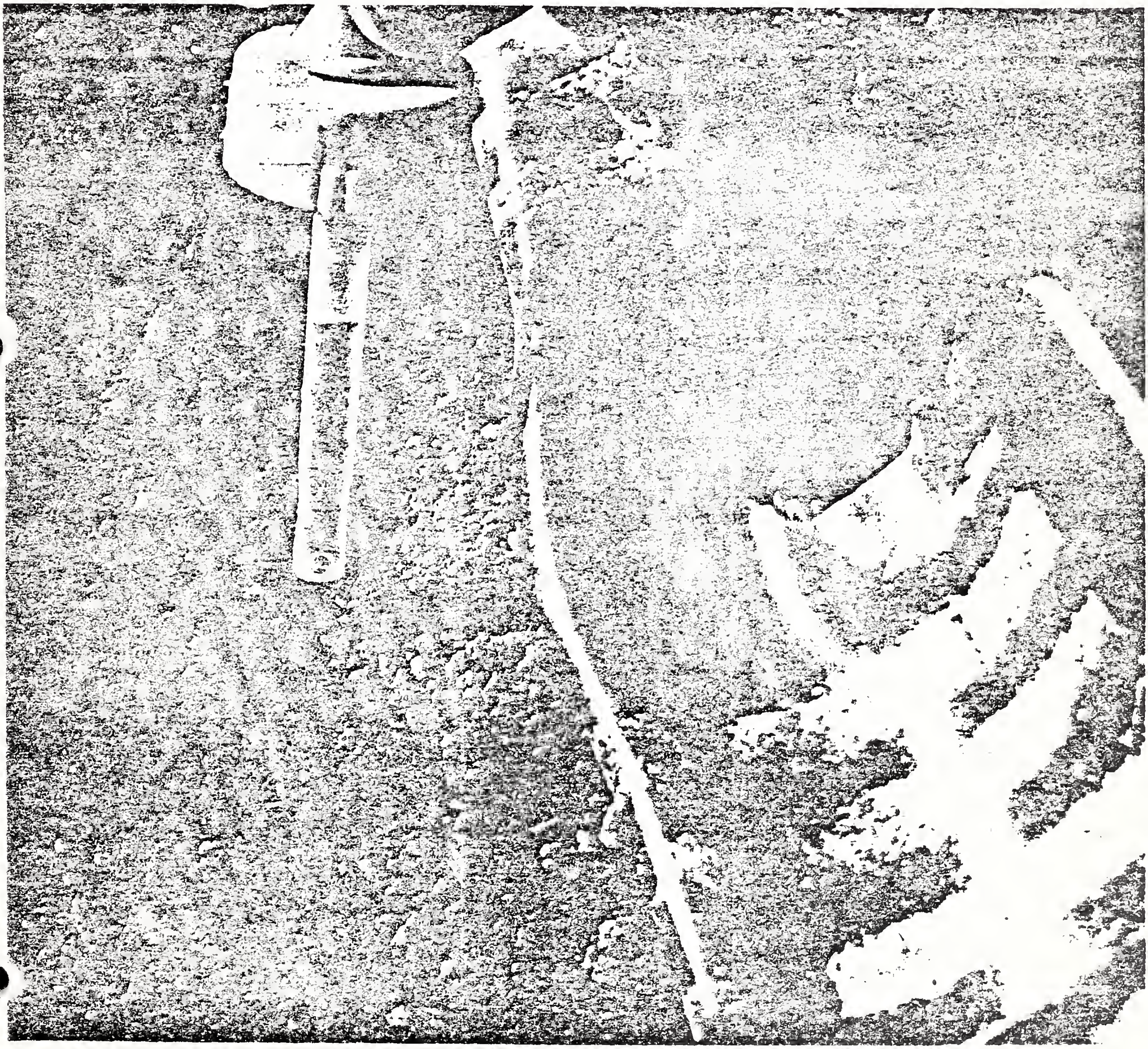
Forest Service

Pacific Southwest
Forest and Range
Experiment Station

General Technical
Report PSW-46

Water Repellent Soils: a state-of-the-art

Leonard F. DeBano



IV. What we don't know for sure or at all

A. Cause of apparent flow changes in some large streams

B. Effects of logging on snowmelt during rainfall

C. Occurrence of extreme precipitation

D. Extent of soil wettability problems

E. Extent of fog zones

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Effects of Logging and Associated Activities

on Hydrologic Processes

Dennis Harr

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B. New information

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EFFECTS OF TIMBER HARVEST ON STREAMFLOW IN THE RAIN-DOMINATED PORTION OF THE PACIFIC NORTHWEST

by

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Proc., Workshop on Scheduling Timber Harvest for Hydrologic Concerns
November 1979, Portland, Oregon

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CONVERSION TABLE

1 mm	=	0.04 in
1 cm	=	0.39 in
1 m	=	3.28 ft
1 km^2	=	0.386 mi^2 or 247 acres
1 liter/sec·ha	=	10.93 $ft^3/sec \cdot mi^2$

EFFECTS OF TIMBER HARVEST ON STREAMFLOW IN THE RAIN-DOMINATED PORTION OF THE PACIFIC NORTHWEST

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ABSTRACT

Research results from experimental watersheds and plot-level studies in 11 major locations from British Columbia to north coastal California have illustrated the magnitude of changes in streamflow after timber harvest. Annual water yields have increased up to 62 cm, some summer low-flows have quadrupled, and size of peak flows have increased, decreased, or remained unchanged in small headwater basins. Increases in summer flow have disappeared after 4-5 years, and annual water yield increases have diminished as revegetation has proceeded. Although increases in annual water yield and summer flows are probably most readily predicted, neither has any influence on channel erosion processes in the rain-dominated portion of the Pacific Northwest. Our capability of predicting changes in the higher flows directly involved in channel erosion processes is poor and is hindered considerably by our incomplete understanding of subsurface flow of water and runoff production during winter storms, of the mechanics of snowmelt from shallow snowpacks during rainfall, and how each of the foregoing is affected by timber harvest activities.

INTRODUCTION

Each year some 54 million cubic meters (12 billion board feet) of timber are harvested from about 170,000 ha (420,000 acres) of forest land in western Oregon and western Washington alone. Because timber harvest activities have the potential for altering streamflow characteristics in forested watersheds, USDA Forest Service land managers are attempting to schedule harvest in these watersheds over a number of years to minimize the impact on water quality and aquatic ecosystems.

The responsibilities and obligations of the USDA Forest Service in managing National Forest lands have been stated in the Multiple Use-Sustained Yield Act of 1960, the National Environmental Policy Act of 1969, and, more recently, the National Forest Management Act of 1976. Under these Acts the Forest Service is directed to "manage all the

various renewable surface resources of the National Forest...without impairment of the productivity of the land...to provide for methods to identify special conditions or situations involving hazards to the various resources," and to "ensure that timber will be harvested from National Forest System lands only where soil, slope, or other watershed conditions will not be irreversibly damaged [or] where protection is provided for streams [streambanks]."

To help the USDA Forest Service fulfill these management obligations requires that everyone responsible for management decisions and actions affecting streamflow have a common and accurate perception of natural forest hydrologic systems and how they are affected by timber harvest activities. Helping to achieve that common understanding of hydrologic systems is a major purpose of not only this paper but also the workshop of which this paper is a part. The objectives of this paper are: (1) to review our understanding of the function of rain-dominated hydrologic systems, (2) to discuss what research has learned about the effects of timber harvest activities on these hydrologic systems, and (3) to examine our capability of predicting effects of harvesting activities on the quantity and timing of streamflow.

Many of my examples and much of my supporting data will be from watershed studies conducted by the Pacific Northwest Forest and Range Experiment Station. These data are most familiar to me and are probably more extensive than data from other areas.

Before I go further, I would like to qualify the title of this paper. When the workshop organizing group met to decide on workshop agenda and structure, it arbitrarily divided the broad subject of "effects of timber harvest on hydrologic systems" into two parts: (1) snow-dominated systems described by Troendle and Leaf,[1/] and (2) rain-dominated systems. Whether or not a hydrologic system is rain or snow-dominated probably is best illustrated by the distribution of annual runoff relative to annual precipitation. Where annual runoff closely follows annual precipitation, the system clearly is dominated by rain. Many west-side systems fall into this category which is illustrated by the runoff pattern of the North Fork Alsea River in figure 1. Where most precipitation occurs during winter but major runoff does not occur until spring, the system is obviously dominated by snow. Most east-side systems are in this category which is illustrated by the runoff pattern of the Colville River in figure 1.

Of course, all natural systems cannot be categorized so simply. Many watersheds, because they span either a wide elevational range or a relatively narrow range at higher elevations, exhibit a pattern of

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[1/] Troendle, Charles A. and Charles F. Leaf. Effects of timber harvest on water yield and timing of runoff--snow region. Paper presented at Workshop on Scheduling Timber Harvesting for Hydrologic Concerns, Portland, Oreg., November 27-29, 1979.

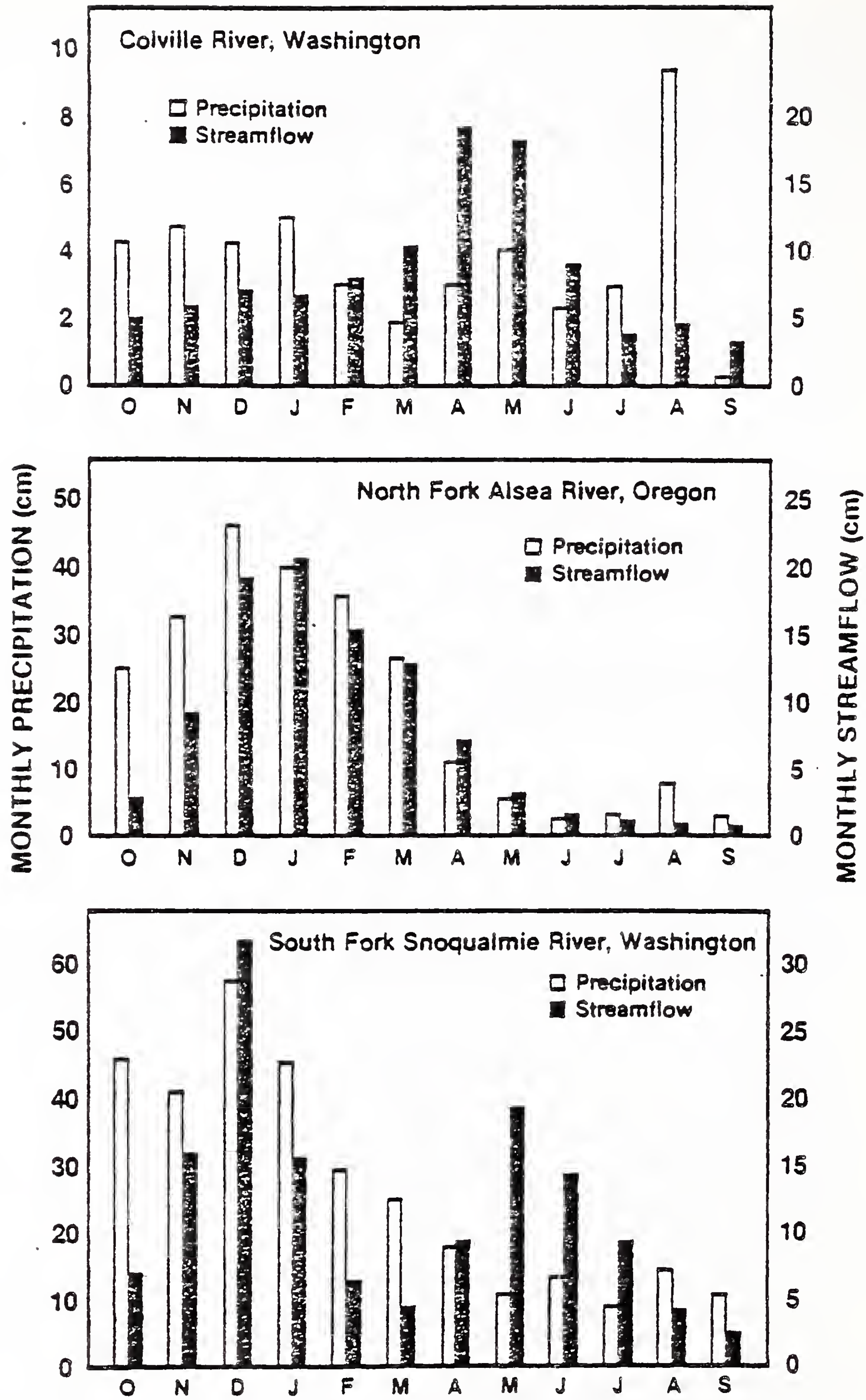

Figure 1—Distribution of annual precipitation and streamflow during 1976 for the Colville, North Fork Alsea, and South Fork Snoqualmie Rivers.

annual runoff with two peaks. The first peak results from fall and early winter rain and the second results from spring melt of winter snow. These watersheds, such as the South Fork Snoqualmie River in figure 1, have been placed in the rain-dominated category, but some of what Troendle and Leaf[1/] described is applicable to these watersheds during spring snowmelt.

The rain-snow dominance question is clouded even further by the rapid melting of shallow snowpacks during rainfall--the so-called rain-on-snow phenomenon. A particular system may fall easily into the rain-dominated category as defined earlier, but several important geomorphic processes--such as soil creep, earthflow, and channel erosion--may be dependent on snowmelt during rainfall although, on the average, less than 5% of total annual precipitation may fall as snow. Thus, in a sense, some parts of apparently rain-dominated systems actually may be dominated by snowmelt, and rain domination may be just a myth of forest hydrology for much of the Pacific Northwest.

For the purposes of this workshop, the rain-dominated hydrologic systems are found from British Columbia to northern California (fig. 2). In Oregon, they are found west of the Cascade Range below about 1500 m elevation and about 1200 m in northern Washington and 1400 m in southern Washington.

RESEARCH HISTORY

Study of the watershed level of natural forest hydrologic systems in western Oregon began in 1946 with the establishment of the Willamette Basin Snow Laboratory in the Blue River drainage by the U.S. Army Corps of Engineers (fig. 2). Soon after the H. J. Andrews Experimental Forest was established in 1948 in the Lookout Creek drainage immediately south of Blue River, parts of it were used for watershed-level experiments by USDA Forest Service Research to determine natural hydrologic characteristics and how they are affected by logging activities. Measurements of streamflow began in three small, low-elevation watersheds (designated HJA-1, 2, 3) in October 1952.

In 1957 USDA Forest Service Research began the Fox Creek Watershed Study in three small basins (designated FC-1, 2, 3) within the city of Portland's Bull Run municipal watershed. Also in October 1957, Oregon State University began the cooperative Alsea Watershed Study in three small watersheds (Needle Branch, Flynn Creek, and Deer Creek) to evaluate effects of roads and clearcut logging on aquatic resources in the Oregon Coast Ranges.

USDA Forest Service watershed management research was expanded in the 1960's to include studies in watersheds in somewhat different climatic and vegetation zones. Hydrometeorological records began in the Coyote Creek watersheds (designated CC-1, 2, 3, 4) in 1964 to determine effects of several silvicultural systems on water resources in the mixed conifer

Figure 2—Approximate boundary of rain-dominated portion of the Pacific Northwest. Also shown are locations of major forest hydrology research areas.

zone of southwestern Oregon. Also in 1964, a study was begun in watersheds HJA-6, 7, and 8 in the H. J. Andrews Experimental Forest high elevation watersheds containing second-growth timber. More recently two small, low-elevation watersheds (HJA-9, 10) became the sites of intensive hydrometeorological and ecological studies as part of the International Biological Program's Coniferous Forest Biome.

With the exception of the Alsea Watershed Study which terminated in 1973, all the other watershed-level studies listed above are still in progress. Results from these active studies have been supplemented by data from the Canadian Forestry Service's Carnation Creek Watershed Study on western Vancouver Island, from studies by the University of British Columbia at Jamison Creek north of Vancouver and at the UBC Research Forest near Haney, from the USDA Forest Service Research's Caspar Creek Experimental Watersheds and the U.S. Geological Survey's Redwood Creek studies in north coastal California, and from studies by the University of Washington on the west side of the Olympic Peninsula in Washington.

THE REGION

In this workshop, we are particularly interested in hydrologic processes that can be sufficiently altered by timber harvest activities to cause changes in the quantity or timing of streamflow. We would also like to understand why various processes vary throughout the region because such variability of hydrologic processes hinders interpretation of research data and makes difficult the extrapolation of research findings to other areas for the purpose of accurately predicting the effects of timber harvest activities on quantity and timing of streamflow.

The climate of the Pacific Northwest is characterized by heavy fall and winter precipitation and relatively dry spring and summer periods. Winter weather is dominated by a westerly flow of moist air with a procession of fronts and low pressure areas; 75-85% of annual precipitation generally occurs between October 1 and March 31 (fig. 3). Less precipitation occurs in April and May, and during the June-September period the cyclonic activity characteristic of the winter is greatly reduced. This annual pattern is controlled by the seasonal position of the north Pacific subtropical anticyclone. In winter its southerly position allows storms to enter the Pacific Northwest, but in summer its northerly migration pushes storm tracks to the north. The general climate of the Pacific Northwest is characterized further by mild temperatures with prolonged cloudy periods, muted annual temperature extremes, and relatively narrow diurnal temperature fluctuations. Winters are cool with average January temperatures of 2-5°C (35-41°F) and average minimum January temperatures of -2-2°C (28-35°F) over most of the region. July temperatures average 15-22°C (59-72°F) with average maximum July temperatures of 20-32°C (68-90°F).

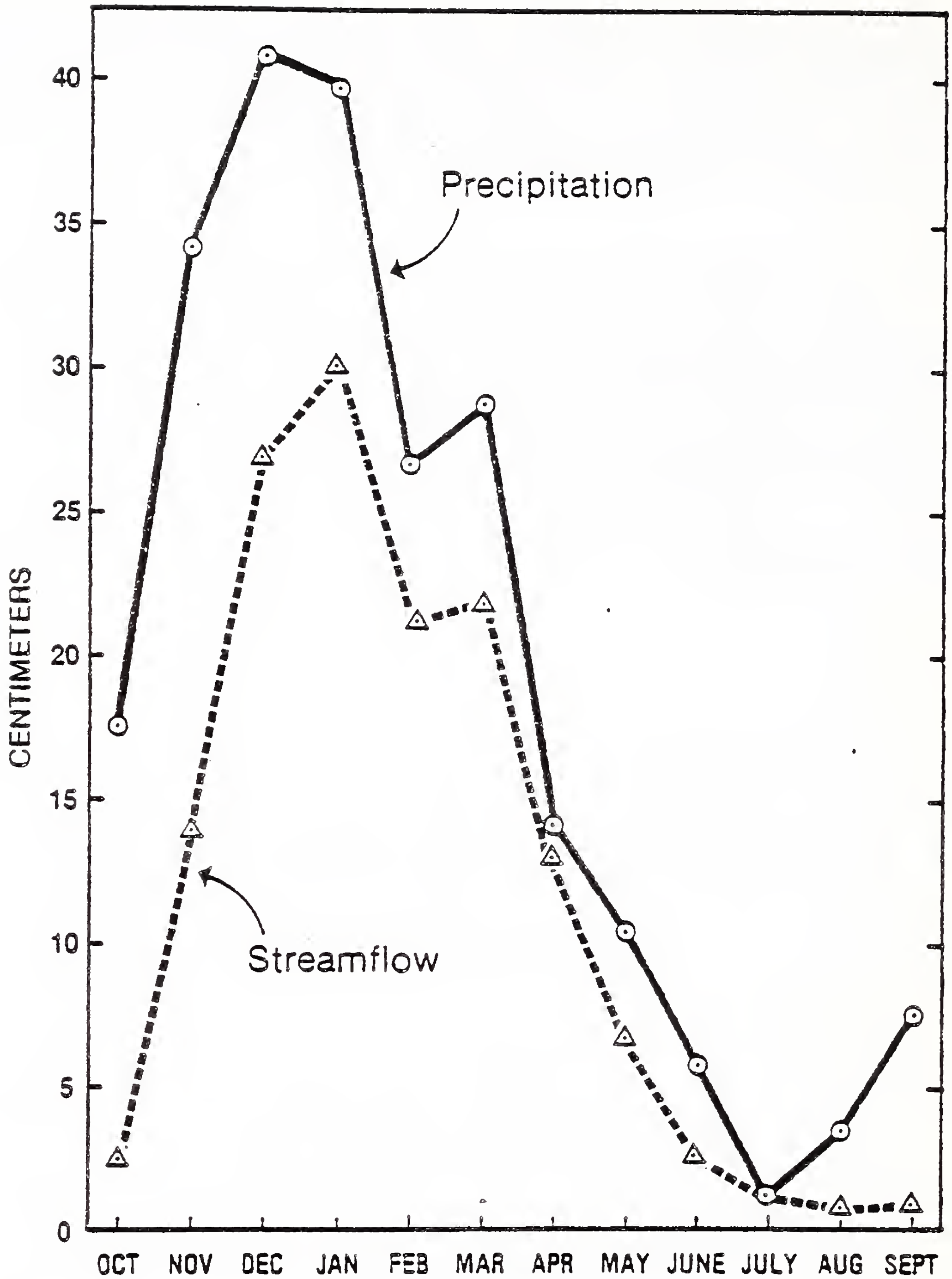

Figure 3—Mean monthly precipitation and streamflow at watershed HJA-2, H. J. Andrews Experimental Forest, Oregon, 1953-77.

Generally speaking, the systems we have classified as rain-dominated receive annual precipitation ranging from about 100 cm along the margins of both the Willamette Valley in Oregon and the Puget Trough in Washingt to more than 300 cm along the windward side of the Cascades, the Olympic

Table 1—Average annual precipitation and 24-hr precipitation for selected stations in western Washington and Oregon

Station	Elevation	Average annual precipitation[1/]	24-hr precipitation[2/] for return periods of 2 yr	5 yr	25 yr
	m	cm			
Washington:					
Forks	107	296	14	17	22
Quinault	67	338	15	19	24
Quilcene	37	139	10	12	15
Shelton	7	163	10	12	17
Darrington	168	204	10	13	17
Randle	274	154	7	9	12
Wind River	350	256	11	14	18
Oregon:					
Astoria	2	168	9	11	14
Newport	47	180	8	10	13
Valsetz	352	321	17	19	24
Powers	70	159	12	14	17
Estacada	125	151	9	11	14
Oakridge	389	117	6	9	11
Sexton Summit	1 170	93	8	10	13

[1/] National Oceanic and Atmospheric Administration (1977a, 1977b).

[2/] Miller et al. (1973a, 1973b)

and the Oregon Coast Ranges (table 1). At upper elevations, snowpack depth may reach 150 cm in some years but generally is 60-90 cm. At elevations between 400 m and 1000 m in western Oregon and 350 m and 650 m in western Washington, snowpacks are transient in most years; they rarely remain longer than 1-2 weeks and usually melt in 3-4 days during rainfall.

Precipitation is characterized by long duration (12-72 hr) storms of low to moderate intensity (generally <6 mm/hr) as illustrated by 24-hr precipitation amounts with return periods of 2-5 yr (table 1). According to the precipitation-frequency atlases for Washington and Oregon (Miller et al. 1973a 1973b) average intensity during 6-hr precipitation with a 25-yr return period ranges from 9 to 12 mm/hr over most of the region, although some local average intensities may be more than 18 mm/hr.

For the most part, forest lands in western Washington and Oregon are characterized by steep, extensively dissected topography. In the Olympic and northern Cascade Mountains of Washington, glaciation has

strongly influenced many landforms; major river valleys are broad and U shaped, with steep sideslopes. On the other hand, glaciation has been negligible in most of rain-dominated western Oregon. Downcutting by streams and subsequent mass wasting of oversteepened sideslopes has created rugged topography with steep slopes and knifelike ridges. These conditions are particularly characteristic of the Coast Ranges, the Klamath Mountains, and the middle portion of the Western Cascades in Oregon. Drainage densities during the winter rainy season are high over most of the rain-dominated region of the Pacific Northwest, in some cases over 6 km/km^2. Steep slopes and high drainage densities greatly influence the movement of water through a watershed. In general, watershed responses to changes in rates of water input are directly related to slope steepness and drainage density.

Soils in the region have derived from a variety of parent materials, but texture of most surface horizons tends to be loamy, ranging from clay loams to sandy and gravelly loams. Some local areas, however, do exhibit clay surface horizons. Because of cementing of primary soil particles by organic matter and other agents, surface soils have large amounts of secondary porosity and wide ranges of pore sizes. Subsurface horizons are generally less permeable because of lower porosity, particularly in the macropore size range. The soil profile as a whole not only is capable of accepting and rapidly transmitting rain and snowmelt water, but also can store 30-40 cm in its top 1.2 m of depth.

The so-called rain-dominated region of the Pacific Northwest contains dense forests which represent the maximum development of temperate coniferous forests in the world. Forest vegetation influences several portions of the forest hydrologic system within certain limitations imposed by climate and physiography as will be described in the next section.

HYDROLOGIC PROCESSES

In this section I will review the various processes of the natural forest hydrologic system to identify processes that can be altered by timber harvest activities. Such a review will aid in the subsequent discussion of changes in streamflow which have been observed after logging, as well as in the review of our capability of predicting changes in streamflow throughout the rain-dominated portion of the Pacific Northwest.

INTERCEPTION

Interception of precipitation has been measured most extensively in the old-growth forests of Douglas-fir. At the Willamette Basin Snow Laboratory annual interception averaged 46 cm during a 4-yr period when annual precipitation averaged 325 cm (U.S. Army Corps of Engineers, 1956). Near Watershed HJA-2 in the H. J. Andrews Experimental Forest, Rothacher (1963) measured 46 cm of interception when annual

precipitation was 213 cm during a 1-yr study under 61-m high trees with average canopy density of 89%. Rothacher also found that stemflow on old-growth Douglas-fir and hemlock had a negigible effect on net precipitation. Krygier (1971) measured 28 cm of interception when annual precipitation was 100 cm in a study under a stand of second-growth Douglas-fir 37 m high with an average canopy density of 99%.

Amount of rain interception loss is dependent on storm size. Rothacher (1963) found very little summer rainfall penetrated the crown canopy until about 1.3 mm of rain had fallen. For summer storms of 25 mm, 50 mm, and 90 mm, interception losses were 5 mm, 10 mm, and 16 mm, respectively. During winter rainy periods, interception losses were also dependent on amount of precipitation but were relatively less than in summer. During storms of 50-100 mm, interception loss averaged about 12% (6-12 mm); for storms of 100-150 mm, it averaged 7% (7-10 mm); and for storms greater than 200 mm, slightly more than 4% (9 m).

Where it has been measured, snow interception has not differed greatly from rain interception. At the Willamette Basin Snow Laboratory snow interception in water equivalent averaged 22 cm compared with 24 cm of rain interception. Snow interception is dependent not only on storm characteristics, such as amount of snow and air temperature, but also on melt conditions immediately after snowfall and on branching habit of forest trees (Smith 1974).

In some local areas, interception of clouds or fog may increase precipitation. For example, Isaac (1946) found annual precipitation owing to fog interception and drip under the forest canopy near the Oregon coast was 252 cm, 52 cm more than in the open. In the Bull Run Municipal Watershed near Portland cloud drip may add significantly to annual precipitation under the forest canopy.[2/]

INFILTRATION

One of the most important characteristics of the forest hydrologic systems in the Pacific Northwest is the capability of surface soil to accept rain and snowmelt water. The result is that overland flow is extremely rare under natural conditions and is confined to intermittent stream channels, bare rock, areas of extremely shallow soil, or frozen soil. Many surface soils in the Pacific Northwest can accept more than a hundred times more water than they are likely to receive from precipitation and snowmelt. For example, Dyrness (1969) reported percolation rates of more than 500 cm/hr for surface horizons of soil on watersheds HJA-1, 2, and 3 in the H. J. Andrews Experimental Forest.

[2/] Harr, R. Dennis. Streamflow after patch-cut logging in small drainages within the Bull Run municipal watershed, Oregon. Unpublished manuscript on file at Forestry Sciences Laboratory, Corvallis, Oregon.

Working with soil from watershed HJA-10, Ranken (1974) measured saturated hydraulic conductivities in excess of 800 cm/hr. Yee (1975) measured conductivities in excess of 300 cm/hr for soil of the Oregon Coast Ranges.

SUBSURFACE FLOW

Once in the soil, water is subject to gravitational and capillary forces that cause it to move and frictional forces that tend to restrict movement. Because of the steep slope of most forest land and because soil permeability generally decreases with depth, water begins to move downslope as it also moves deeper into the soil. The nature of this downslope movement is one of the most poorly understood processes in forest hydrology.

At least two types of subsurface flow exist on forested slopes in the Pacific Northwest. One type consists of flow through the soil matrix by a displacement process described by Hewlett and Hibbert (1967), in which water stored in the soil is displaced by water from precipitation. The second type consists of flow through interconnected soil channels extending from the forest floor through mineral soil. This second type has been described in detail for other parts of the United States by Whipkey (1969) and Aubertin (1971).

Water may move through the soil matrix either as unsaturated flow or saturated flow. In studies by Yee and Harr (1977) and Harr (1977) unsaturated flow dominated although occasional saturated zones were detected. In the latter study, saturation occurred in the soil mantle where there was an abrupt decrease in vertical permeability associated with a reduction in relative amount of macropores. Although other less abrupt decreases in macroporosity with soil depth did not cause saturation, they were instrumental in giving soil water movement a sizable downslope component. Results of these two studies generally support the displacement theory of Hewlett and Hibbert (1967). Between storms, soil profiles remained wet so that they were able to respond quickly to subsequent rainfall; even 1 meter below the forest floor, soil water movement increased greatly during rainfall.

Indirect evidence of flow through interconnected channels has been obtained in forested watersheds in south-coastal British Columbia. Because hydrologic response of soil at 70 cm and near bedrock was faster than horizons closer to the forest floor, Chamberlin (1972) concluded that open drainage routes must exist between the surface organic layer and basal soil horizons. Cheng et al. (1975) also found that water reached the B horizon before it reached horizons closer to the surface, indicating nonuniform wetting or wetting of soil from below. Water apparently was transported rapidly through interconnected channels to lower horizons. Flow through such channels may also occur elsewhere in the Pacific Northwest.

EVAPOTRANSPIRATION

Several measurements have been made of daily evapotranspiration from Douglas-fir forests of various heights in the Pacific Northwest using the energy budget approach (McNaughton and Black 1973, Gay and Holbo 1974), but no measurements of annual evapotranspiration are available. Because so little energy is available for evaporation from soil under a dense forest canopy, most all evapotranspiration consists of only transpiration. Estimates of annual actual evapotranspiration (excluding interception loss) have been about 40-50 cm (U.S. Army Corps of Engineers 1956, Luchin 1973).

Estimates of evapotranspiration can be obtained from long-term measurements of precipitation and streamflow in forested watersheds. At watershed HJA-2 in the H. J. Andrews Experimental Forest, for example, annual precipitation averaged 230 cm during the 1953-78 period, and annual streamflow averaged 142 cm. This leaves 88 cm for evapotranspiration which, in this case, includes interception loss. If 40-45 cm is subtracted for interception, the remainder of 43-48 cm represents largely transpiration, quite similar to the 42 cm estimated at the Willamette Basin Snow Laboratory using the Thornthwaite method (U.S. Army Corps of Engineers 1956). At Coyote Creek watershed CC-4, evapotranspiration plus interception loss averaged 61 cm during the 1966-76 period (Harr et al. 1979). Because overstory canopy density here is less than half of that where Rothacher (1963) conducted his interception study and because there is less annual precipitation at Coyote Creek, interception loss is estimated to be only 20-25 cm. Thus, evapotranspiration at Coyote Creek is estimated to be 36-41 cm. At Needle Branch in the Alsea Watershed Study, evapotranspiration plus interception loss averaged 59 cm during the 7-yr prelogging period and about 43 cm at the nearby Flynn Creek and Deer Creek watersheds. Because Flynn Creek and Needle Branch watersheds were forested primarily with red alder which is leafless during the time of fall and winter rains, annual interception in these watersheds was probably much less than that measured in old-growth Douglas-fir by Rothacher (1963). I estimate that interception loss was 15-20 cm and evapotranspiration was 30-45 cm in these coastal watersheds.

STREAMFLOW

For any time period, streamflow is the difference between precipitation and losses to interception and evapotranspiration plus or minus any change in soil water storage. On an annual basis, streamflow amounts are generally high (table 2) owing to high annual precipitation in the Pacific Northwest. Highest annual streamflows are found on the Olympic Peninsula and lowest annual streamflows are found in southwestern Oregon.

Most small headwater streams in the region respond quickly to precipitation (fig. 4). Soon after rainfall begins, streamflow begins to increase and the source area of streamflow expands and contracts

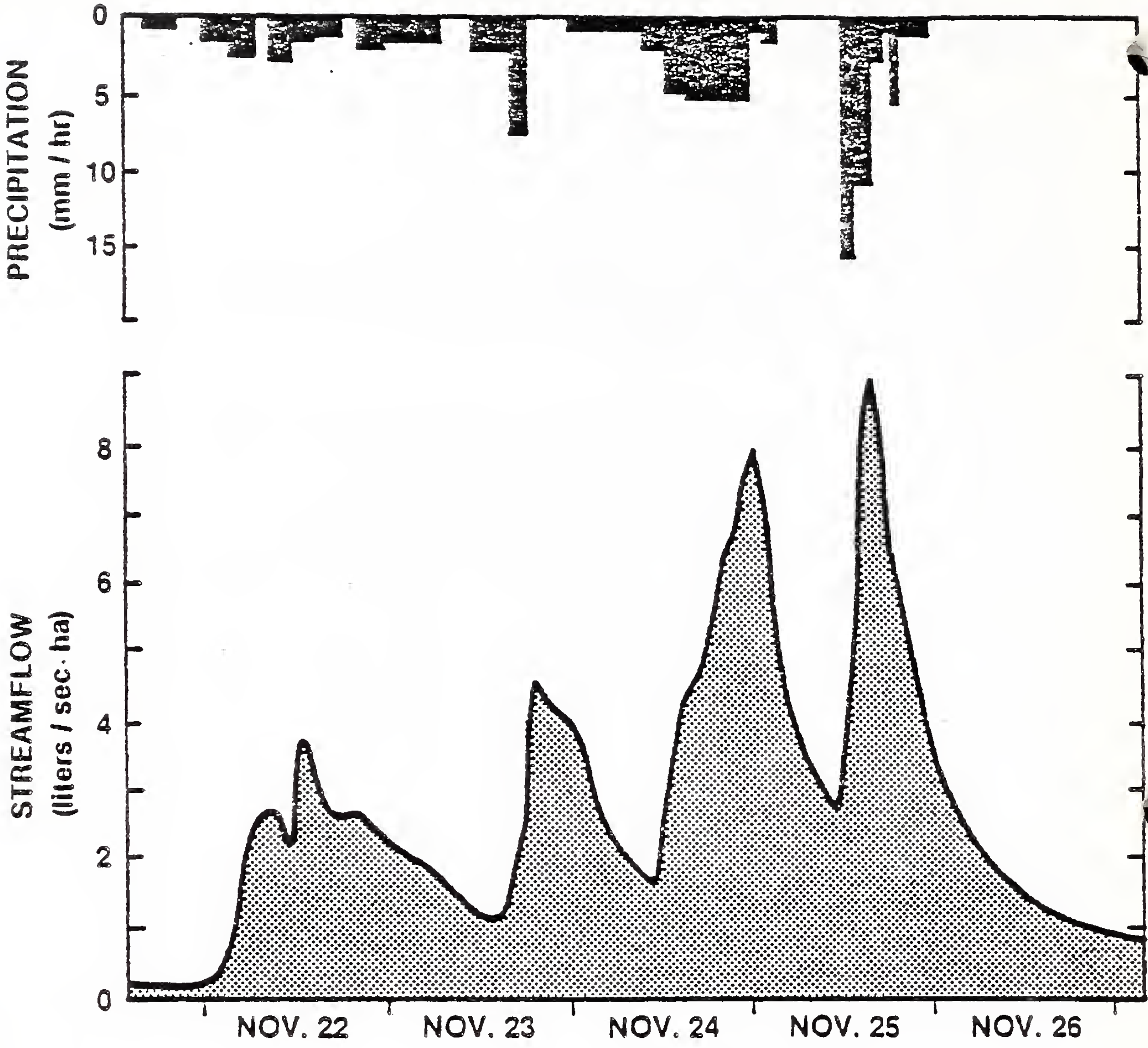

Figure 4—Precipitation and streamflow at watershed HJA-9, H. J. Andrews Experimental Forest, Oregon, November 21-26, 1977.

according to rainfall characteristics and the capability of the soil mantle to store and transmit water (Harr 1976b). This phenomenon is called the variable source area of streamflow (fig. 5). Initially, storm runoff results from channel interception and rainfall on the wet areas adjacent to stream channels. If a storm continues, an increasingly greater proportion of the watershed contributes to storm runoff, and eventually the channel network grows to many times its minimum perennial dimensions. As perennial streams lengthen and flow appears in intermittent channels, drainage density increases, and the watershed becomes more efficient in producing runoff. If rainfall continues at a relatively high rate, streamflow will also continue at a

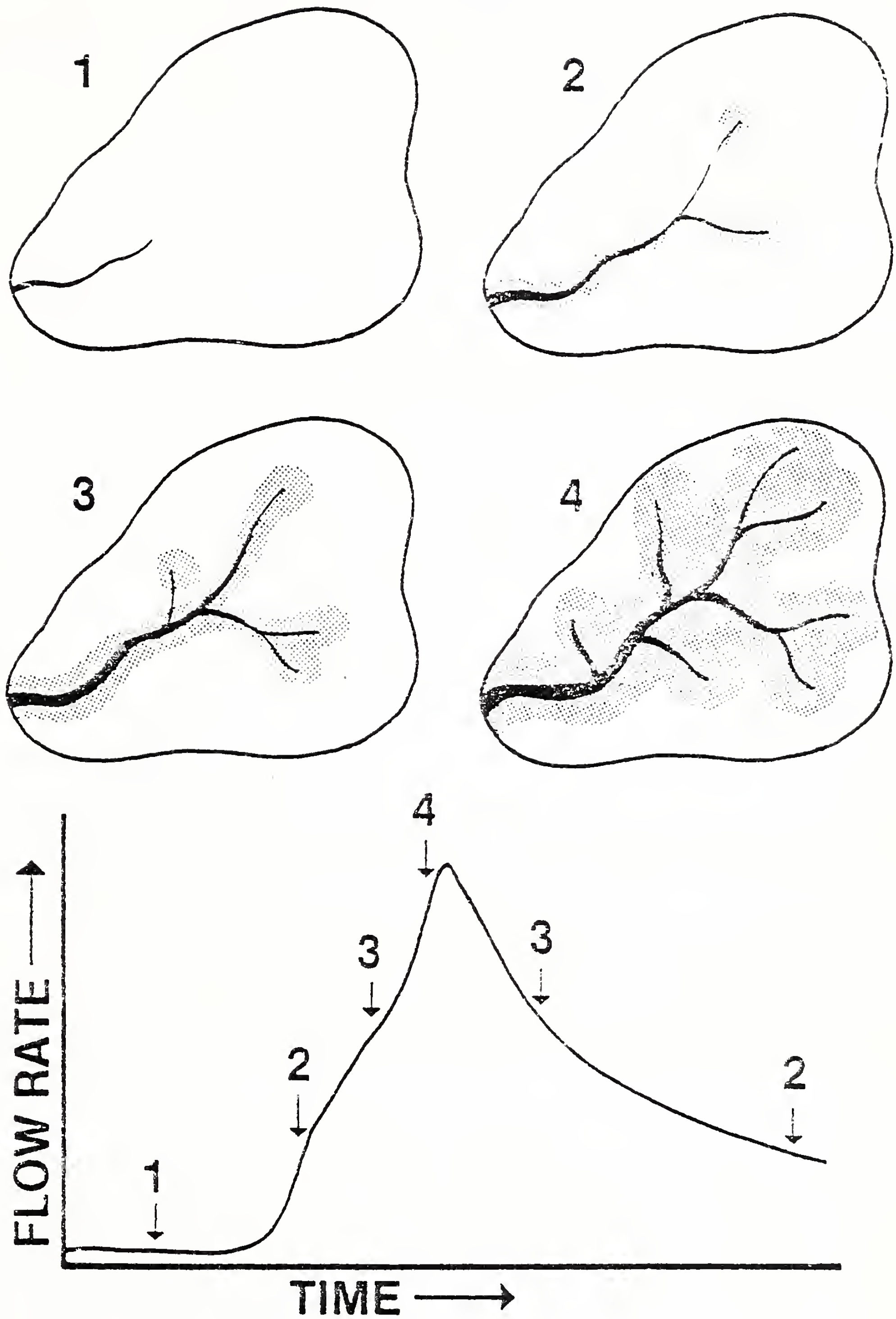

Figure 5—Time-lapse view of a watershed with variable source area of streamflow. The time of each source area condition relative to storm runoff is shown by the location of numbers on the storm hydrograph.

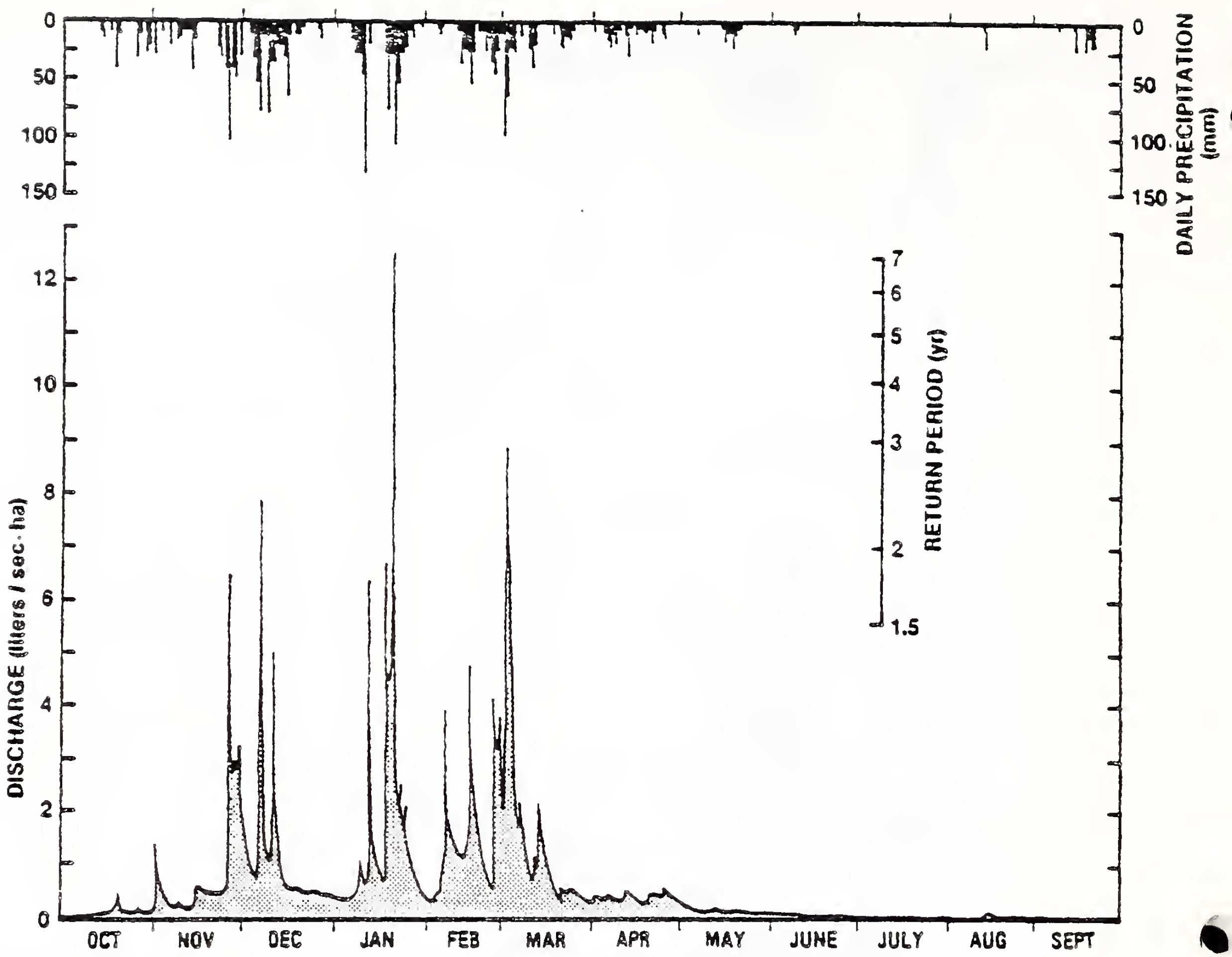

Figure 6—Daily precipitation and instantaneous streamflow at watershed HJA-2, H. J. Andrews Experimental Forest, Oregon, 1972 water year.

high rate. Conversely, if rainfall rate decreases or rainfall ceases altogether, then streamflow will peak almost immediately (fig. 4) and then begin to decrease rapidly as permeable soils on steep slopes drain quickly (Harr 1977).

Over the course of the rainy season, the sequence described above may occur 10-20 times or more, depending on the frequency of storms entering the region from the Pacific Ocean. In 1972 at HJA-2, for example, there were 16 storms which deposited at least 50 mm of precipitation (fig. 6). Eight of these caused peak discharges >4 liters/sec.ha, and three caused peak discharges greater than the estimated mean annual peak of 7.5 liters/sec·ha. Between stormflow periods, discharge decreased to only 2-4 liters/sec·ha, depending on length of time soil was allowed to drain in the absence of rainfall.

Because of steep sideslopes, highly permeable soils, and the rapid

response of streams to changes in rainfall rate, high rates of streamflow are short lived. This is illustrated by the flow duration curve (fig. 7) constructed from the streamflow data plotted in figure 6. During the 1972 water year, flows greater than the estimated mean annual peak occurred less than 1% of the time in watershed HJA-2. About half the time, flow was less than the 0.3 liters/sec·ha baseflow which occurred between storm runoff periods in winter.

Maximum flows of nearly all streams in the Pacific Northwest have resulted from rapid snowmelt during prolonged, heavy rainfall. In western Washington streams, maximum flows generally have occurred in November or December (table 2). In western Oregon, maximum flows have occurred most frequently in December and January.

The timing of minimum flows of record has been much more variable, ranging from August to December for the streams listed in table 2. In most years, minimum flows occur between mid-July and early September. Flow ceases in many small headwater basins during the summer dry period.

Dividing maximum flows of record by minimum flows of record (table 2) shows the range of flows that can be experienced at one location in the Pacific Northwest. For large basins, maximum flows of record are up to several thousand times larger than minimum flows of record. The range is much greater for small streams. At watershed HJA-2 in the H. J. Andrews Experimental Forest, maximum flow is more than 17,000 times greater than minimum flow of record. Of course, in any one year ratios of maximum to minimum flows are less than those derived from table 2;

Table 2—Mean annual, maximum, and minimum streamflows for selected watersheds of western Washington and Oregon through the 1978 water year

Stream	Drainage area	Length of record	Mean annual yield[1]	Maximum streamflow of record		Minimum streamflow of record	
	km^2	yr	mm	liters/sec·ha	month	liters/sec·ha x 10^{-3}	month
Washington:							
Wynoochee River	192	52	3 387	34.8	Dec.	84	Sept.
Duckabush River	172	39	2 159	14.8	Nov.[2]	74	Sept.
Snoqualmie River	971	47	2 399	17.8	Nov.[2]	2.8	Aug.
South Fork Cedar River	15.5	35	2 208	42.7	Dec.[2]	35	Nov.
Stetattle Creek	57	46	2 885	42.6	Nov.[2]	45	Nov.
Oregon:							
Siletz River	523	58	2 695	22.2	Nov.	26	Sept.
Alsea River	865	38	1 585	13.6	Dec.	15	Sept.
Flynn Creek	2.03	15[3]	1 949	19.2	Jan.	15	Oct.
South Santiam River	451	42	1 637	17.3	Dec.	14	Dec.
Lookout Creek	62	20	1 850	30.5	Dec.	29	Nov.
HJA-2	0.603	26	1 410	16.6	Dec.	0.95	Sept.
HJA-8	0.214	15	1 295	19.2	Dec.[2]	1.3	Sept.-Oct.
HJA-9	0.085	10	1 311	14.1	Jan.	3.3	Aug.-Sept.
South Umpqua River	1 163	39	801	14.6	Dec.	4.9	Sept.
CC-4	0.486	13	578	14.4	Dec.	2.9	Aug.

[1] Mean annual yield expressed as a uniform depth over entire watershed area.

[2] Streamflow frequently has a secondary annual peak during spring snowmelt.

[3] Streamflow measurement discontinued at the end of the 1973 water year.

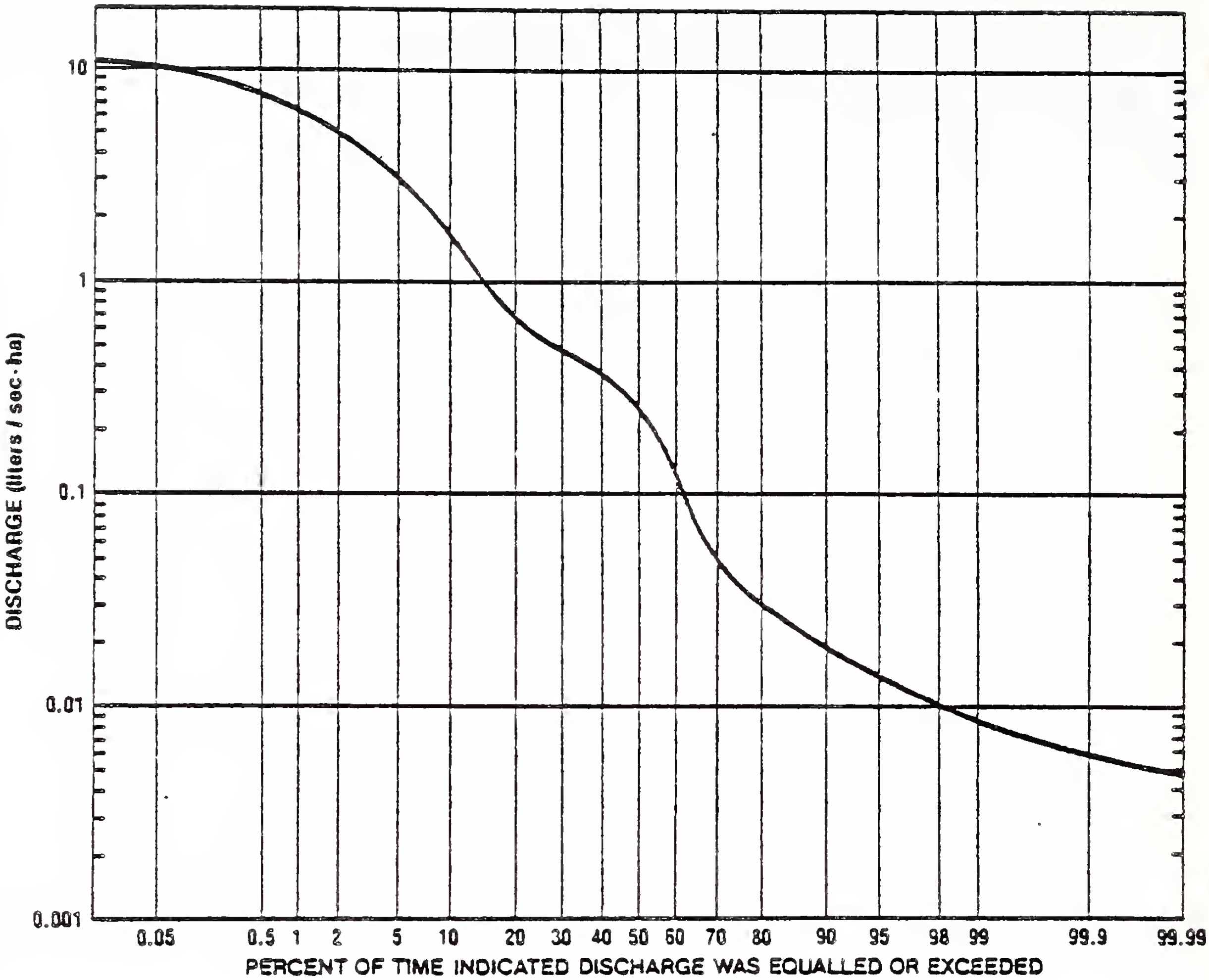

Figure 7—Duration curve of streamflow at watershed HJA-2, H. J. Andrews Experimental Forest, Oregon, 1972 water year.

for the 1972 water year shown in figure 6 the ratio is 1495 to 1. Thus, little can be gained toward understanding winter flow conditions simply by observing stream channels during the summer when drainage densities are at their lowest.

SNOWMELT

Because maximum streamflows in the so-called rain-dominated portion of the Pacific Northwest are usually caused by melting of shallow snowpacks during rainfall, a brief review of snowmelt processes should be helpful at this point. There are several major differences in snow hydrology between western Washington and Oregon and other parts of the United States.

Snowpacks in western Oregon and Washington, like those in the Sierra

Nevada of California, are classified as "warm" in contrast to the "cold" snowpacks of the central Rocky Mountains or the Northeast (Smith 1974). A warm snowpack's interior temperatures remain at or near 0°C throughout the pack's existence. This temperature is hydrologically important because relatively little energy is required to initiate melting. A warm pack can yield water quickly during a period of high air temperature, rainfall, or both, once the pack's storage capacity for liquid water has been satisfied. In many instances, snowpacks at lower elevations (350–1000 m) of western Washington and Oregon are shallow enough to be melted completely during rainstorms.

The heat transfer processes described by Troendle and Leaf[1/] operate to melt snow in western Washington and Oregon as they do elsewhere. The relative importance of these various processes constitutes the second major difference in snow hydrology in this region. Incoming shortwave radiation, the major source of energy for melt in most of the United States, is a minor source of energy for melt during rainfall in western Washington and Oregon. According to the U.S. Army Corps of Engineers (1960), snowmelt during rainfall (commonly referred to as rain-on-snow) is a special situation for which certain simplifying assumptions can be made in the snowmelt equation so melt can be estimated by several indices listed in table 3.

The relative importances of melt resulting from the various sources of energy are shown graphically in figure 8 for 24-hr average air temperatures (T_a) of 2°C and 10°C. At T_a = 10°C (fig. 8B) and P_r = 8 cm, convection-condensation melt (M_{ce})—i.e., melt resulting from warm air moving across the snow surface and from heat released as water vapor condenses on the snow surface—is the major component of total melt; 45% of total melt results from convection-condensation. Second is

Table 3--Snowmelt indices for 24-hr melt during rainfall under forest conditions (adapted from U.S. Army Corps of Engineers 1956)

Source of melt	Equation for 24-hr melt[1/]
Short-wave radiation	M_{rs} = 0.18 cm/day
Ground heat	M_g = 0.05 cm/day
Long-wave radiation	M_{rl} = 0.133 T_a cm/day
Convection-condensation	M_{ce} = 0.206 T_a cm/day
Rainfall heat	M_p = 0.0126 P_r T_a cm/day

[1/] T_a = average 24-hr air temperature (°C); P_r = 24-hr rainfall (cm).

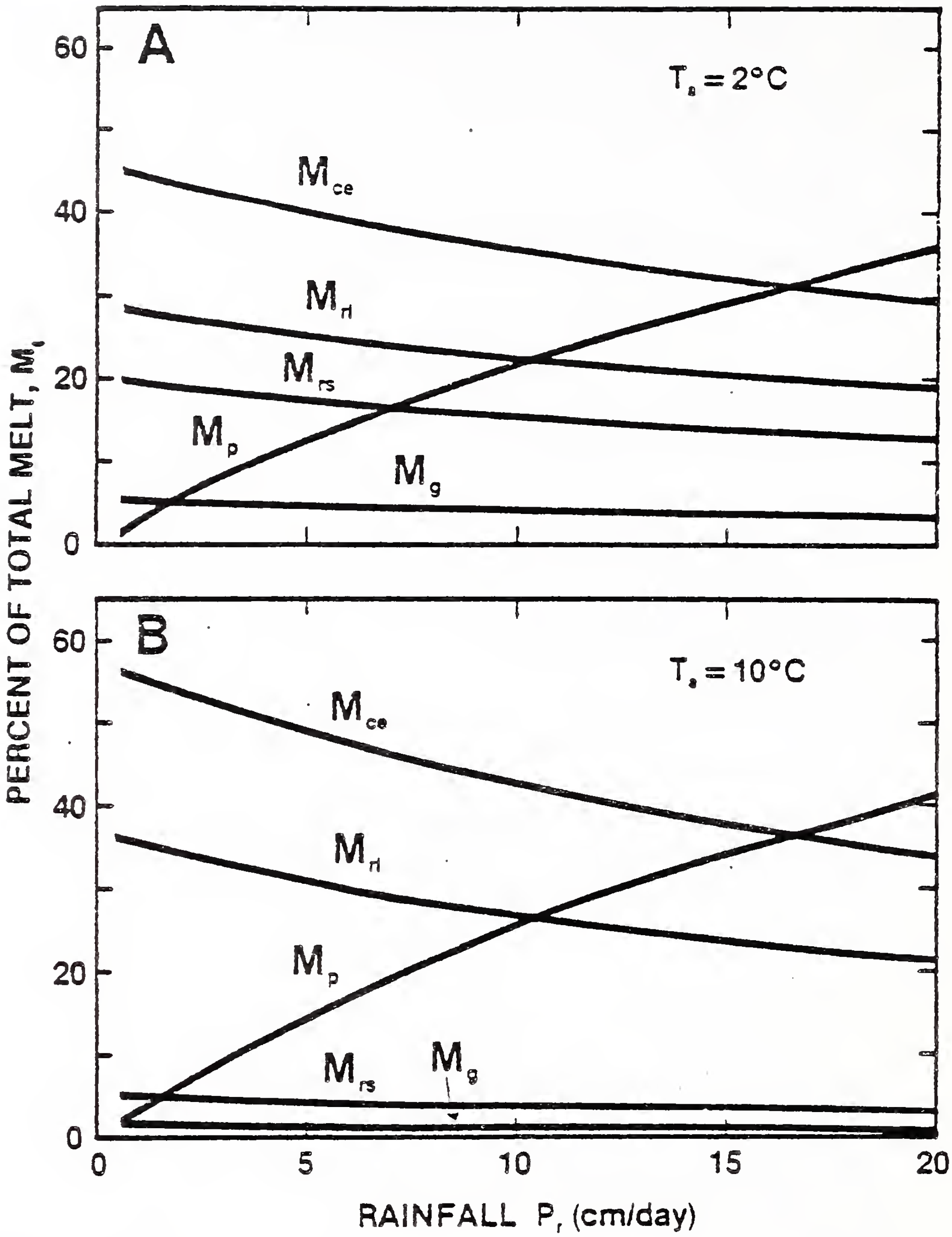

Figure 8—Proportion of total snowmelt caused by various components of melt during rainfall at mean daily air temperatures of 2°C and 10°C. Melt components are defined in table 3 (adapted from U.S. Army Corps of Engineers 1956).

long-wave radiation melt (M_{rl}) followed by melt caused by heat contained in rain (M_p). Although the phrase "rain-on-snow" implies snow is melted by warm rain, this is not entirely the case. Warm rain is the greatest

source of energy for melt only when 24-hr rainfall (P_r) is more than 17 cm. This is a very important point because such rainfall rates are infrequent over much of the region; for most conditions, rain is only the third greatest source of energy for snowmelt. Shortwave radiation accounts for less than 5% of total melt during rainfall. Of course, figure 8 and the indices given in table 3 cannot be used to estimate snowmelt during sunny periods.

EFFECTS OF TIMBER HARVEST ON STREAMFLOW

Research elsewhere in the United States in the 1930's and 1940's had shown that forest cutting and forest growth could have a major influence on water yield and that the undisturbed forest provides the maximum opportunity for controlling runoff from flood-producing storms. Watershed management research was begun in the Pacific Northwest in the late 1940's to determine the effects of timber harvest on streamflow characteristics under the climatic, physiographic, and vegetative conditions of this region. Questions to be answered: (1) Can logging increase annual water yields? (2) Does logging affect floods? (3) Can the timing of runoff be altered by logging so as to improve the naturally poor annual distribution of streamflow? These questions have been answered to some degree for a few locations in the Pacific Northwest under certain conditions. There have been, however, several misconceptions about what has been learned and how it can be applied to other areas in the Pacific Northwest. And there are other important questions yet to be answered about the effects of logging on streamflow. What follows is a review of research findings concerning the effects of timber harvest on streamflow in the so-called rain-dominated region of the Pacific Northwest.

Much of what is known about the effects of timber harvest on streamflow has come from studies using paired experimental watersheds. One watershed of each pair is the control or unlogged watershed, and the second is the treated (logged or altered in some other way) watershed. For a time before treatment, streamflow characteristics are measured at each watershed. During this calibration period, linear regression is used to develop a relationship for a streamflow characteristic such as annual water yield, minimum flows, and peak flows between the control watershed (dependent variable) and the watershed to be treated (independent variable). After treatment, a new relationship is determined and compared to the calibration relationship, and the difference between relationships is attributed to the treatment.

ANNUAL WATER YIELD

There have been five studies that have illustrated the range of increases in annual water yield after timber harvest in the Pacific Northwest. These studies have utilized 18 experimental watersheds, all of which are located in western Oregon. Characteristics of these

Table 4—Summary of watershed characteristics for watershed-level studies in the Pacific Northwest

Watershed study[1/]	Mean annual precipitation	Mean annual streamflow at control	Type of precipitation	Area	Aspect	Elevation range	Average slope	Parent material	Forest vegetation: Type	Forest vegetation: Age
	cm	cm		ha		m	%			yr
HJA-1, 2, 3	233	141	Rain	60–101	NW	440–1 080	53–63	Altered volcaniclastics	Douglas-fir W. hemlock	300–500
HJA-6, 7, 8	215	129	Rain, snow	13–21	S	830–1 100	27–31	Relatively unaltered volcaniclastics	Douglas-fir	120–130
HJA-9, 10	233	165	Rain	8.5–10	SW	425–715	65–70	Altered volcaniclastics	Douglas-fir W. hemlock	300–500
FC-1, 2, 3	272	175	Rain, snow	59–253	W–NW	860–1 070	5–9	Igneous glacial till	Douglas-fir W. hemlock	300–500
CC-1, 2, 3, 4	133	58	Rain	49–69	N–NE	730–1 065	23–36	Altered volcaniclastics	Douglas-fir Mixed conifer	100–300
Alsea (AL-1, 2, 3)	243	196	Rain	70–303	S	135–485	34–40	Sandstone	Alder Douglas-fir	120
Jamison Creek (JC)	391	—[2/]	Rain, snow	298	SE	300–1 300	48	Igneous glacial till	W. red cedar W. hemlock Douglas-fir	300–500
UBC-1, 2	229	—[2/]	Rain	23–64	S	145–655	12–20	Igneous glacial till	W. hemlock W. red cedar Douglas-fir	300–500
Caspar Creek (CA-1, 2)	112	—[2/]	Rain	426–508	W–SW	90–365	30	Sandstone	W. hemlock Douglas-fir Redwood	70–90

[1/] HJA = H. J. Andrews Experimental Forest, FC = Fox Creek watersheds, CC = Coyote Creek watersheds, UBC = University of British Columbia Research Forest.

[2/] Not available or not measured.

watersheds, along with other research watersheds in the region, are shown in table 4. Table 5 summarizes timber harvest activities by experimental watershed.

The longest period of postlogging measurement of increases in annual water yields is associated with the study at HJA-1 and HJA-3. Logging began in HJA-1 in late August 1962, but, because of operational problems with the new Wyssen fixed skyline system, logging was not completed until 1966.[3/] Heavy slash was broadcast burned in 1966. In 1965, after the watershed had been 90% clearcut, annual water yield increased nearly 54 cm (fig. 9), only about 60% of the total evapotranspiration and interception losses described earlier. In general, there has been a decreasing trend since 1965 although this trend has been less apparent during the last 8 years. For the entire 1965–78 period 68% ($r^2 = 0.68$) of the variation in yield increases is accounted for by the equation,

$$Y = 52.2 - 2.05X_1; \qquad (1)$$

where X_1 is the number of years after logging, and the year 1966 = 1, 1967 = 2, etc. In other words, 68% of total variation in yield

[3/] Mention of commercial names does not constitute an endorsement by the U.S. Department of Agriculture.

Table 5—Summary of timber harvest activity by watershed

Watershed[1]	Activity (see key at right)[2]	Key		
HJA-1	CC, CY-100, BB-100	UC	-	Uncut
HJA-2	UC, UB	CC	-	Clearcut
HJA-3	R, PC-30, CY-30, BB-30	PC	-	Patchcut
HJA-6	R, CC, CY-93, TY-7, BB-100	SC	-	Shelterwood cut (percent basal area removed)
HJA-7	SC-60, CY-40, TY-60, BB-100	R	-	Permanent haul road
HJA-8	UC, UB	CY	-	Yerding by a cable system
HJA-9	UC, UB	TY	-	Yarding by tractor
HJA-10	CC, CY-100, UB	UB	-	Residue unburned
AL-1[3]	R, CC,[4] CY-72, TY-10, BB-82	BB	-	Residue broadcast burned
AL-2	UC, UB	CPB	-	Residue piled by cable and burned
AL-3	R, PC-25, BB-8	TPB	-	Residue piled by tractor and burned
AL-32[5]	R, PC-30, CY-30, UB			
AL-33[5]	R, PC-65, CY-65, UB			
AL-34[5]	CC-90, CY-90, UB			
FC-1	R, PC-25, BB-25			
FC-2	R, UC, UB			
FC-3	PC-25, CY-19, TY-6, UB			
CC-1	R, SC-50, UB			
CC-2	R, PC-30, CY-16, TY-14, CPB-16, TPB-14			
CC-3	R, CC, CY-77, TY-23, CPB-77, TPB-23			
CC-4	UC, UB			
UBC-1	R, CC-75, CY,[6], TY,[6], UB			
UBC-2	UC, UB			
CA-1	R, CC[5], TY-100, UB			
CA-2	UC, UB			

[1] HJA = H. J. Andrews Experimental Forest, FC = Fox Creek watersheds, CC = Coyote Creek watersheds, UBC = University of British Columbia Research Forest, CA = Caspar Creek watersheds.

[2] Except where noted, numbers refer to percentages of watershed area where activities were carried out.

[3] AL-1, 2, 3 refer to Needle Branch, Flynn Creek, and Deer Creek watersheds, respectively.

[4] Only 82% clearcut during study. Additional 18% in the headwaters of Needle Branch was logged in the early 1950's.

[5] AL-32, AL-33, and AL-34 are subwatersheds of AL-3.

[6] Percentages of watershed yarded by cable and tractor are not available.

[7] Selectively logged in three stages over 3 years. At the end of the period, the watershed was in a clearcut condition.

increases is related to time since logging, a gross index of revegetation and increasing water loss through interception and transpiration.

The effect of annual precipitation on size of yield increases is also apparent in figure 9. In general, wetter years exhibited higher increases in yield and vice versa. Indeed, if annual precipitation is taken into account the predictive equation or model becomes:

$$Y = 31.41 - 2.08X_1 + 0.091X_2; \qquad (2)$$

where X_2 is annual precipitation. Adding annual precipitation to the model accounts for a statistically significant portion of total

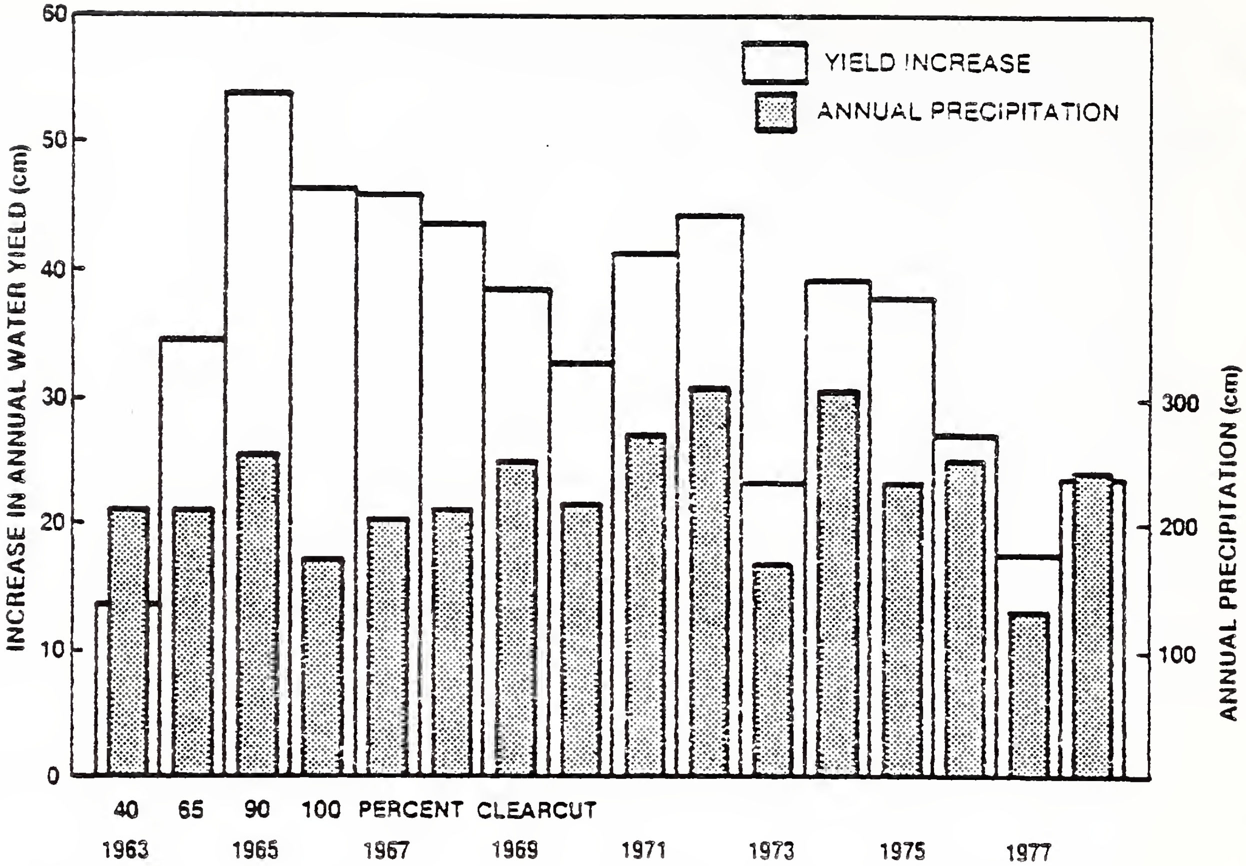

Figure 9—Annual precipitation and increases in annual water yield at watershed HJA-1, H. J. Andrews Experimental Forest, Oregon.

variation in yield increases. The model described by equation 2 accounts for 87% of total variation in yield increases at HJA-1.

Yield increases for all clearcut experimental watersheds are shown in figure 10. Largest increases were noted at AL-1 (Needle Branch) in the Oregon Coast Ranges (Harris 1977) where water yields increased more than 60 cm 3 and 5 years after logging. Yield increases at AL-1 appear also to be influenced by annual precipitation, but, because the Alsea Watershed Study terminated in 1973, the strength of this influence could not be determined. (An undetermined amount of this yield increase is due to road drainage water that flowed into AL-1 from an adjacent logged area.) Why maximum increases at AL-1 and HJA-10 did not occur until the 3rd year after clearcutting is not known.

Yield increases have been smaller at patch-cut HJA-3 than at clearcut HJA-1 because the watershed was altered less by timber harvest (fig. 11). During road construction in 1958, 8% of watershed HJA-3 was cleared. After patch-cutting in three units in late 1962, cleared area totaled 30% of watershed area. Logged units were burned in September 1963. Water yield increases at HJA-3 show a general decreasing trend

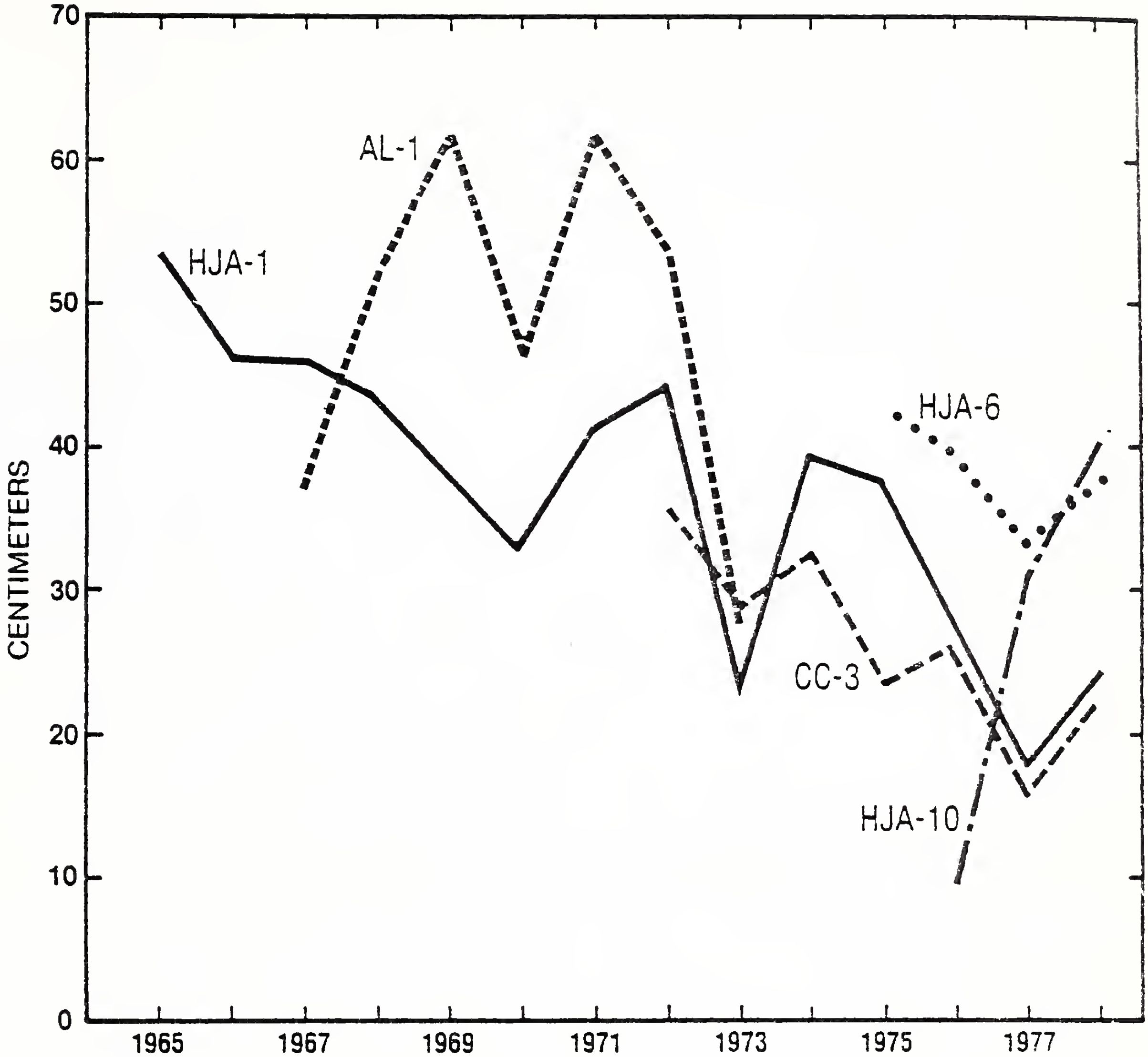

Figure 10—Increases in annual water yield at five clearcut watersheds in western Oregon. Part of the increases at AL-1 are due to road drainage water flowing into the watershed from an adjacent logged area. See tables 4 and 5 for watershed descriptions.

over time but a significant relationship with time was not found. As at HJA-1, greatest increases have tended to occur during wettest years, although this tendency is less apparent at HJA-3 than at HJA-1. Also, at HJA-3 annual precipitation did not account for a significant portion of total variation in annual water yields as it did at HJA-1.

Yield increases for all partially logged experimental watersheds are shown in figure 12. HJA-3, AL-3, CC-1, and CC-2 all contain roads, HJA-3, CC-1, and CC-2 were patchcut, and CC-1 and HJA-7 were logged with

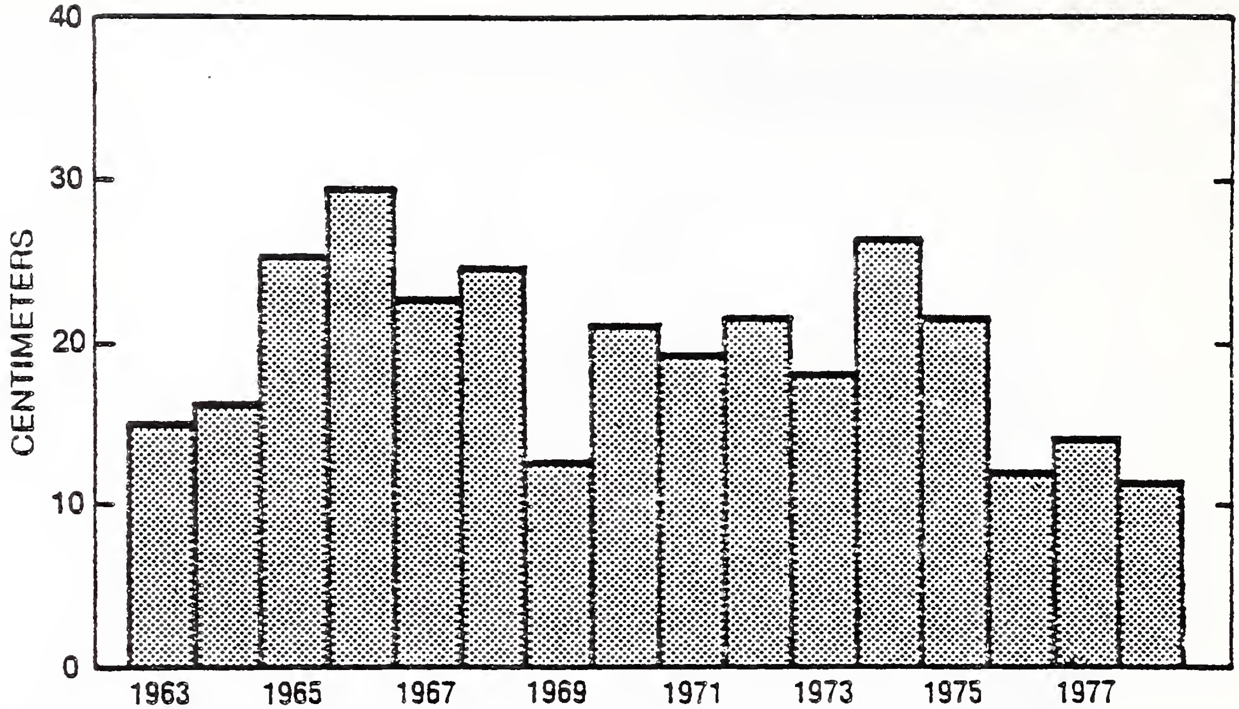

Figure 11—Increases in annual water yield after patch-cutting in watershed HJA-3, H. J. Andrews Experimental Forest, Oregon.

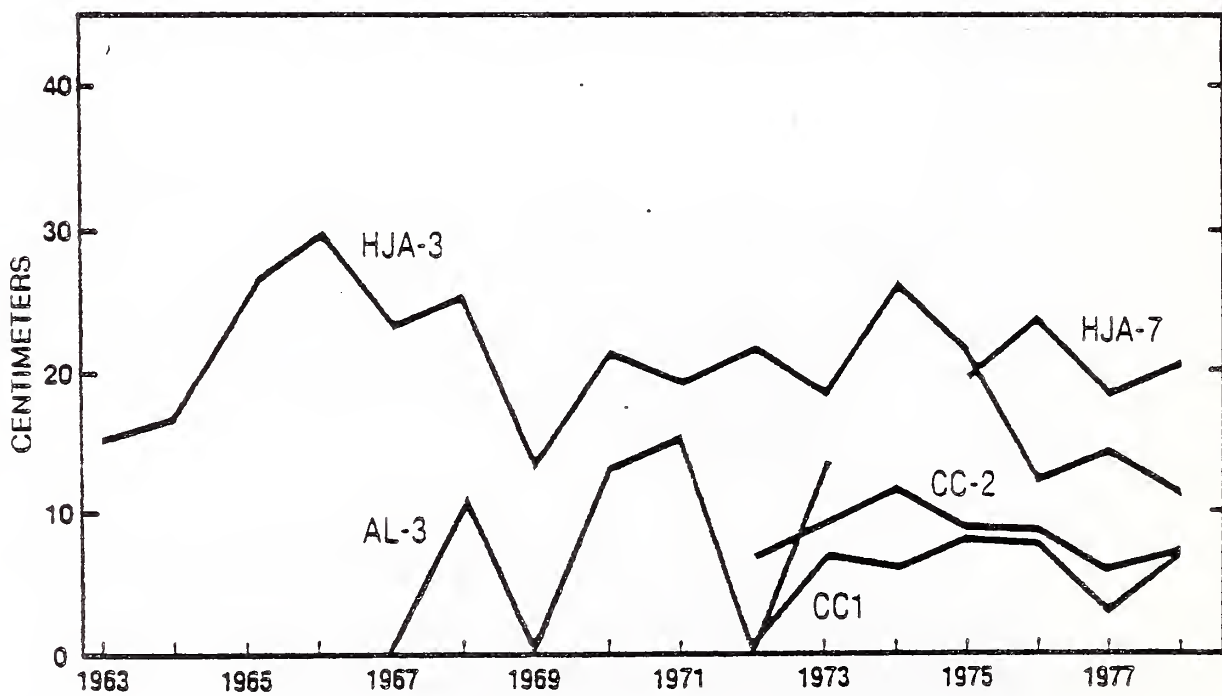

Figure 12—Increases in annual water yield at five partially logged watersheds in western Oregon. See tables 4 and 5 for watershed descriptions.

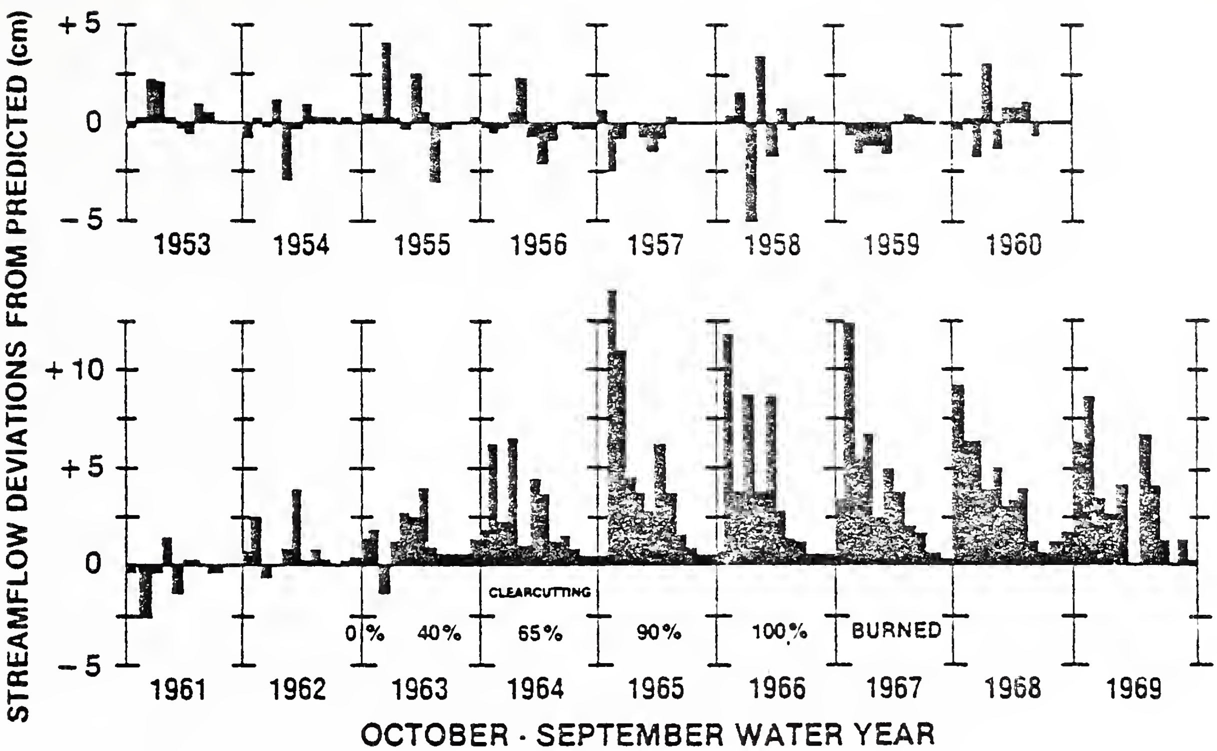

Figure 13--Changes in monthly water yield after logging in watershed HJA-1, H. J. Andrews Experimental Forest, Oregon (Rothacher 1971).

a shelterwood cut. Not included in figure 12 are data from FC-1 and FC-3, patch-cut watersheds where annual water yields decreased slightly though not significantly after logging (see footnote 2). Yield decreases (or at least lack of expected increases) at FC-1 and FC-3 are thought to be related to reduced fog interception after logging.

Seasonal analyses of yield increases have been made only at HJA-1 and 3 and CC-1, 2, and 3 (Rothacher 1970, Harr et al. 1979). At both locations most of each year's increase in water yield occurred during the October-March rainy season (figs. 13-14). Part of this rainy season increase is due to reduced transpiration in a clearcut during the growing season. Wetter soils at the end of the growing season require fewer fall rains for recharging soil water and are able to yield more water to streamflow. Another portion of the rainy season increase in yield is due to reduced interception loss after removal of forest vegetation.

Implications of Water Yield Increases

Certain characteristics of water yield increases combine to make such increases in headwater basins of little consequence downstream. First, increases tend to diminish over time so that only a fraction of a

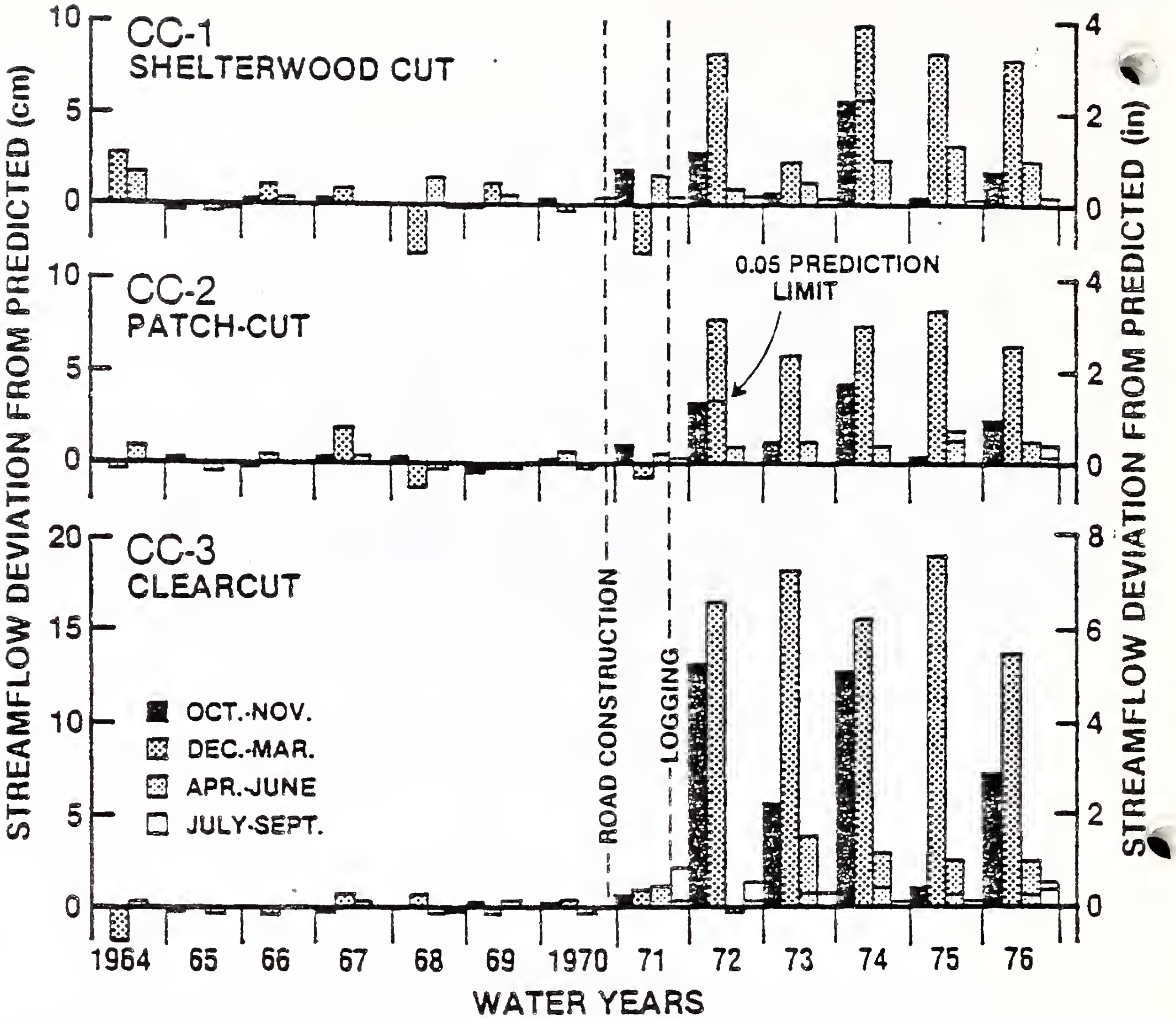

Figure 14—Changes in seasonal water yields after road construction and logging in the Coyote Creek experimental watersheds, Oregon (Harr et al. 1979).

watershed managed for sustained production of timber products will be in a condition to yield appreciably more water. The remainder will yield normal or nearly normal flows which tend to mask increased flows from freshly logged upland areas. Examples used by Rothacher (1970), Bethlahmy (1974), and Harr et al. (1979) show that forest cutting in large watersheds—about 100 km^2—probably would be only about 4-6%, an amount well within the normal accuracy of streamflow measurement for the large watershed. Management under nonsustained yield of timber products, however, could result in somewhat greater water yields for a large watershed.

A second characteristic of water yield increases that bears on the value

or utility of increases downstream is their timing. Substantial increases during the fall-winter rainy season will do little to satisfy summer demand for water. This timing, coupled with the fact that water yield increases tend to occur during wet years, further limits the real benefits commonly attributed to increased water yield after timber cutting. To use these increases would require storage facilities, and if storage facilities were present, storage of yield increases would be of miniscule importance compared with storage of normal winter runoff for release during the summer dry period.

Perhaps the major implications of water yield increases demonstrated by these watershed studies lie in the area of erosion by soil creep and earthflow processes. Water yield increases are caused by, and are indices of, higher soil water contents from reduced interception and transpiration. Most of the downslope movement of soil in creep and earthflow terrain occurs during the fall and winter when maximum soil water contents occur (Swanson and Swanston 1977). Reduced interception and transpiration after logging, by increasing the amount of water entering the soil and decreasing its rate of withdrawal from the soil, may cause higher soil water contents that may cause higher rates of slope movement or longer periods of time soil water contents remain conducive for soil creep or earthflow. Prolonged periods of active creep or earthflow movement during a single rainy season or reactivation of dormant creep and earthflows may be the result (Swanston and Swanson 1976). The effects of such landslides can be felt far downstream as well as in the upland areas where the landslides occur.

In forested watersheds of western Oregon and Washington there is no relationship between increases in annual yield and increases in size of peak flows. In a recent summary of changes in streamflow after logging in western Oregon, yield increases and size of peak flow increases appeared to be independent of one another (Harr 1976a). Thus, changes in annual water yield are of no value in predicting changes in maximum flows in this region.

LOW FLOWS

Although yield increases in absolute terms are greatest during fall and winter months, greatest increases in relative terms have occurred during the summer. At HJA-1, for example, measured summer low flows were four times greater than predicted the first 2 years after logging and three times greater than predicted the year after slash was burned (fig. 15). Because of rapid growth of alder, willow, and other riparian vegetation, increases in yield largely disappeared within 2-3 years. Since 1974, 7 years after slash burning, measured summer flows have been consistently slightly less than flows predicted by the calibration regression equation.

At Coyote Creek, relative increases in summer low flows were similar to those observed during the first few years after logging at HJA-1 (Harr et al. 1979). At CC-3, measured summer flow was three times greater

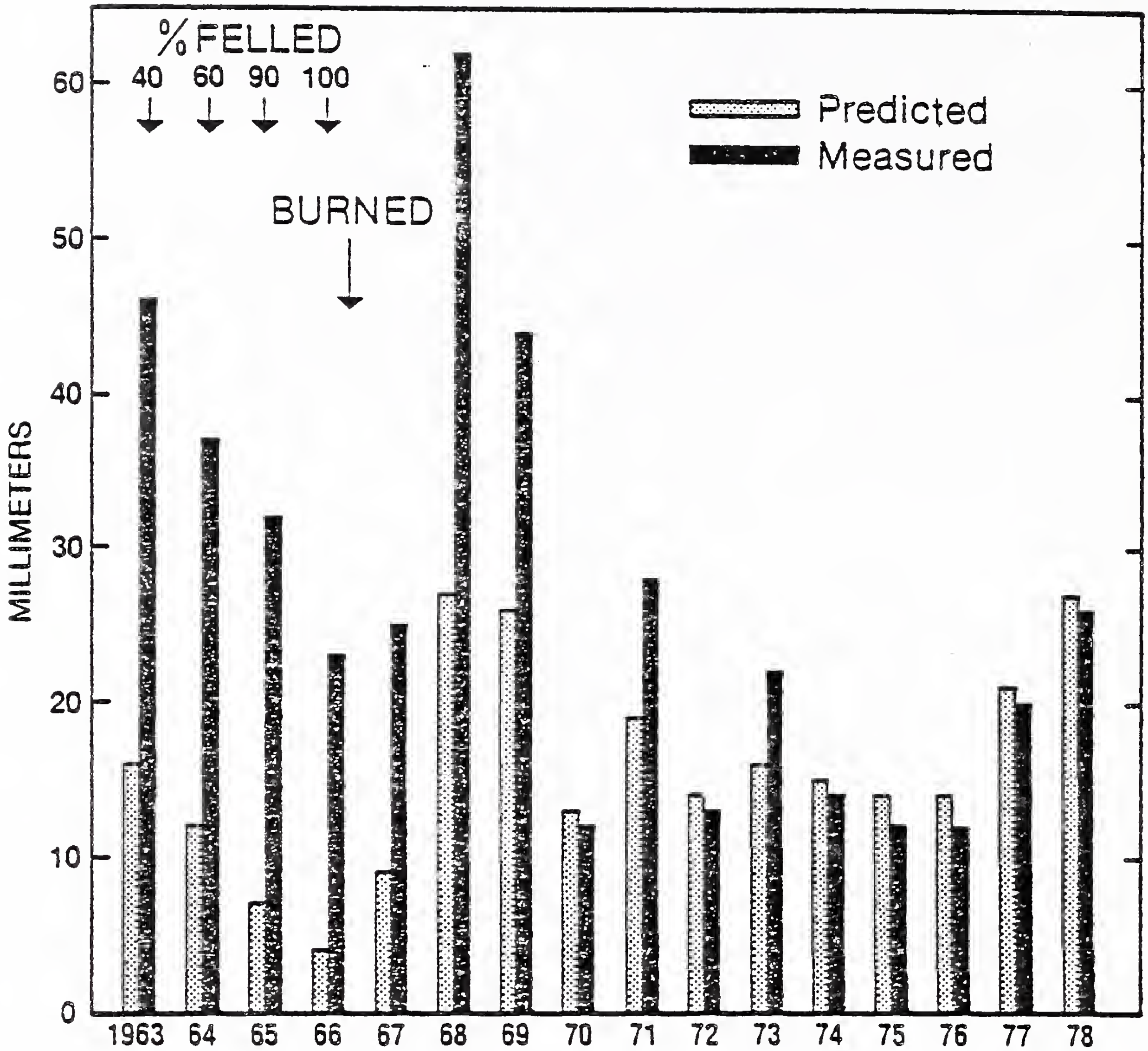

Figure 15—Increases in July-September streamflow at watershed HJA-1, H. J. Andrews Experimental Forest, Oregon.

than predicted the 1st year after clearcutting but, because of rapid growth of alder, willow, and other riparian vegetation, relative size of summer increases have diminished quickly here also. In 1977 and 1978, relative increases in summer flow were only 4% and 16%, respectively. At CC-1 and CC-2, the partially logged watersheds nearby, relative increases in summer flows have been generally much smaller than at CC-3.

In the Alsea Watershed Study in the Oregon Coast Ranges, the number of low-flow days (i.e., days flow was below 0.11 liter/sec·ha) decreased (low flows increased) after AL-1 was 82% clearcut and burned (Harr and Krygier 1972). The effect of patch-cutting 25% of AL-3 on low-flow days

was less pronounced. Since this study was discontinued in 1973, longevity of summer increases is unknown.

Summer low flows have not increased after logging in all watershed studies. At FC-1 and FC-3 in Portland, Oregon's Bull Run municipal watershed, summer low flows were significantly reduced following patch-cut logging (see footnote 2). Reduction in low flows tentatively has been attributed to reduced fog drip from late spring to early fall.

PEAK FLOWS

For several decades there have been controversy and speculation about the effects of timber harvest on floods in the Pacific Northwest. Resultant property damage from flooding during 1977 and 1978 in western Washington did not lessen the controversy. The first analysis of the effects of logging on maximum flows in the Pacific Northwest was published in 1959, although concern about these effects had begun long before. After analyzing peak flow data for several large watersheds in western Oregon, Anderson and Hobba (1959) concluded that logging had increased the size of peak flows caused by rain alone and by rain with snowmelt.

Results from a number of studies on experimental watersheds in the Pacific Northwest in recent years suggest that a simple generalization cannot be made about the effects of timber harvest on peak flows. Although research results may appear inconsistent, the change in size of peak flows after logging generally can be explained in terms of what portion of the forest hydrologic system was altered by logging activities and to what degree.

The most common cause of increased size of peak flows has been wetter, hydrologically more responsive soils after timber cutting. Because of wetter soils in logged areas, less rainfall is required to recharge soil water so that more rainfall can be translated into storm runoff. For example, at HJA-1 initial measured peak flows of the fall were up to 200% greater than flows predicted by the prelogging peak flow relationship (Rothacher 1971, 1973). In other studies in the region (Harr et al. 1975, Ziemer[4/]), as well as elsewhere in the United States (Reinhart 1964, Hornbeck 1973), similar larger peak flows have been noted after logging when differences in soil water contents existed.

On the other hand, Rothacher (1973) found the large winter-season peak flows were unaffected by logging activities in watershed HJA-1. After sufficient rainfall had fallen to recharge soil water storage on forested slopes, logged and unlogged areas responded almost identically. Because surface soils are usually only slightly disturbed during yarding

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[4/] Ziemer, Robert R. Influence of roadbuilding and logging on stormflow in small coastal watersheds. Unpublished manuscript on file at Redwood Sciences Laboratory, Arcata, California.

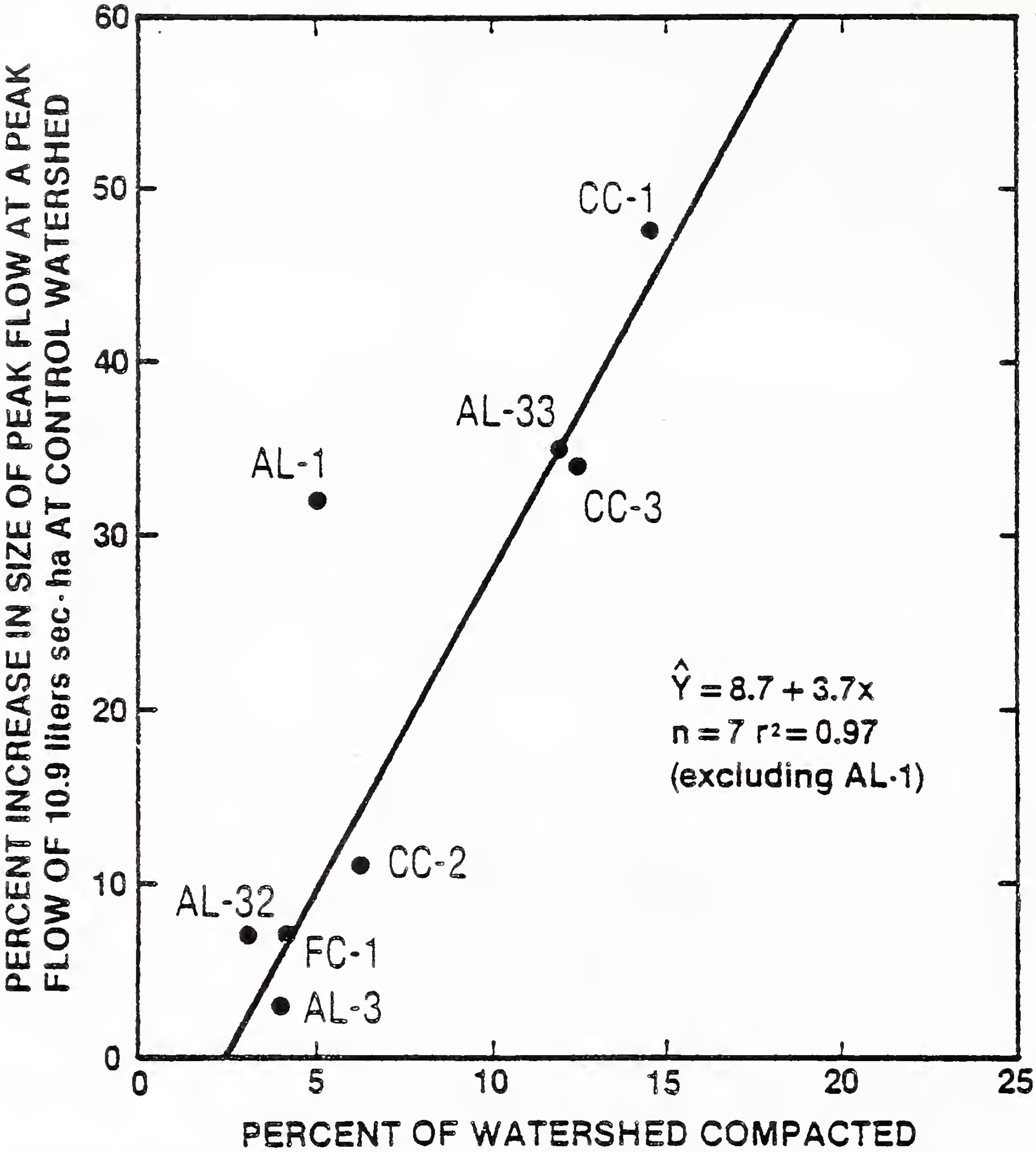

Figure 16—Apparent relationship between soil compaction and increase in size of peak flow. AL-1 has been excluded from the relationship because an undetermined amount of road drainage water flowed into the watershed from an adjacent logged area. See tables 4 and 5 for watershed descriptions.

by cable systems (Dyrness 1965, 1967), soils on HJA-1 were still able to accept all precipitation, and overland flow did not occur.

Winter season peak flows, however, were significantly larger after logging in some watersheds in two other studies in the Pacific Northwest. In the Alsea Watershed Study, Harr et al. (1975) reported larger winter peak flows after logging in AL-33, a small watershed where roads, cutbanks, fillslopes, and landings occupied 12% of total

watershed area. Smaller increases were noted at watersheds with less soil compaction. Winter peak flows were larger after shelterwood cutting in CC-1 and clearcutting in CC-3, two small watersheds in the South Umpqua drainage of southwestern Oregon (Harr et al. 1979). In CC-1, compacted soil from permanent roads, skidroads, and landings occupied nearly 15% of total watershed area, and in CC-3, compacted soil occupied about 12% of total area. At CC-2 where compacted soil occupied only 6% of total watershed area, increases in size of peak flows were proportionately smaller than at either CC-1 or CC-3.

An apparent relationship between soil compaction and peak flows is shown in figure 16. Of course, this relationship is oversimplified because it ignores other factors, such as proximity of compacted areas to streams, continuity of compacted areas so that overland flow can reach streams, interception of subsurface water by road cuts and ditches, and watershed soil and physiographic characteristics. In other words, all areas of compacted soil do not contribute toward increased runoff to the same degree.

In two watershed studies in the Pacific Northwest, peak flows were delayed and reduced in size after timber harvest. At watershed UBC-1 near Haney, British Columbia, soil disturbance during yarding apparently disrupted fast flow through water-transmitting pores and forced water through slower routes in the soil which caused delayed, smaller peak flows after clearcut logging (Cheng et al. 1975). At HJA-10, delay and reduced size of peak flows after clearcut logging were attributed mainly to differences in short-term accumulation and melting of snow (Harr and McCorison 1979). Size of annual (return period of about 2 yr) peak flows caused by rain with snowmelt was reduced 36%. Collectively, peak flows resulting from rain with snowmelt were delayed an average of 12 hr. No significant changes were detected in size or timing of peak flows that resulted from rainfall alone.

Taken collectively, results of watershed studies indicate that size of peak flows may be increased, decreased, or remain unchanged after logging. Whether or not a change occurs depends on what part of the hydrologic system is altered, to what degree, and how permanent the alteration is.

Snowmelt During Rainfall

The potential influences of forests on snowmelt have been known for some time, but actual effects have not been clearly established for mountainous, forested terrain typical of western Washington and Oregon. This is particularly true where snowpacks are shallow and transient during the winter.

According to snow hydrology work done by the U.S. Army Corps of Engineers (1960), clearcut logging could increase the rate of snowmelt during rainfall because turbulent transfer of energy and water vapor to the snow surface would be increased after removal of forest vegetation.

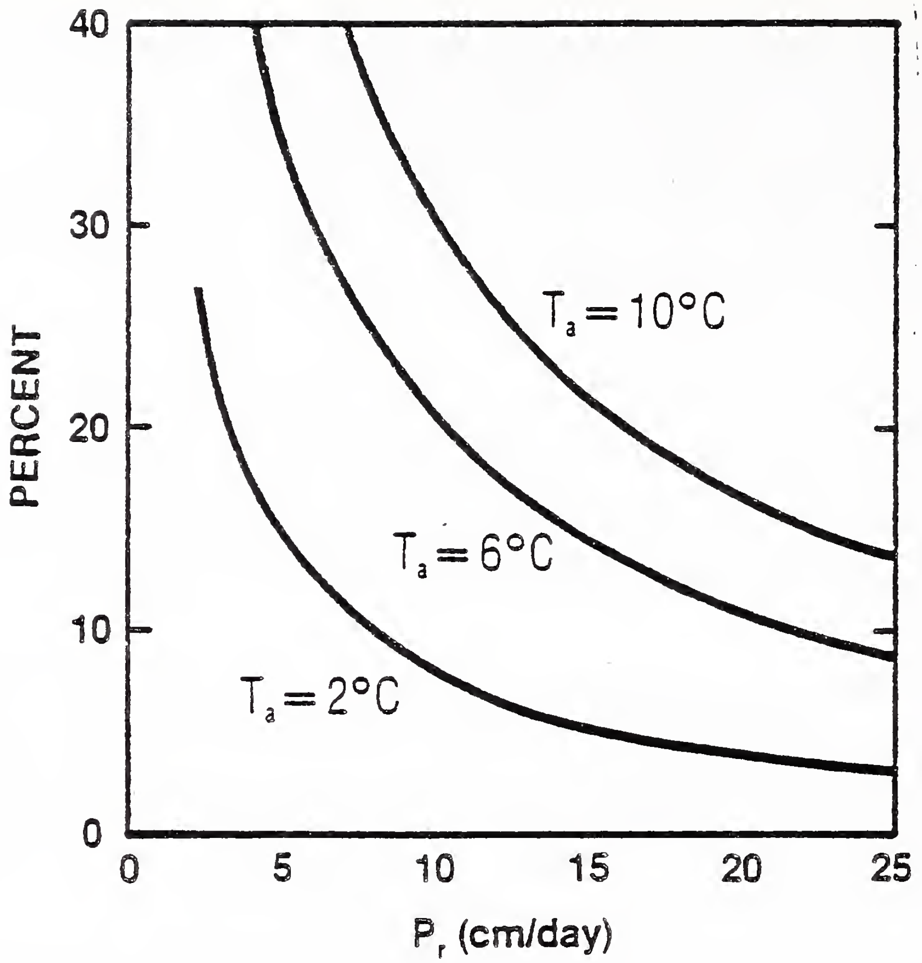

Figure 17—Percent increase in 24-hr water input following timber removal. Increase is a function of average 24-hr air temperature (T_a) and 24-hr rainfall (P_r) and is expressed as a percentage of melt that would have occurred had timber not been removed (adapted from U.S. Army Corps of Engineers 1956).

The equation for convection-condensation melt in table 3 would be replaced by $M_{ce} = 0.086\ V\ T_a$; where V = wind velocity in m/sec 15 m above the snow surface. At a wind velocity of 7.5 m/sec, an average air temperature (T_a) of 6°C, and 24-hr rainfall (P_r) of 10 cm, total melt would be increased about 90% and total 24-hr water input to soil about 20% (fig. 17). In other words, the weather and snowpack conditions which, under forested conditions, would produce a water input event with a return period of about 2 yr, could produce a water input event with a return period of about 10 yr[5/].

[5/] Harr, R. D. Some characteristics and consequences of snowmelt from shallow snowpacks during rainfall in western Oregon. Unpublished manuscript on file at Forestry Sciences Laboratory, Corvallis, Oregon.

At present the effects of timber removal on snowmelt during rainfall are poorly understood and arguments presented here are speculative. Nevertheless, most major runoff and many erosion processes in the Pacific Northwest have been associated with snowmelt during rainfall, and changes in melt under certain circumstances could cause higher runoff than would occur under forested conditions and could adversely affect channel stability. Conversely, depending on weather conditions during snow accumulation and melt, clearcutting may reduce the size of some peak flows as was observed in watershed HJA-10 (Harr and McCorison 1979).

That a potential increase in runoff exists because of changes in snowmelt during rainfall after logging does not mean that all stream channel segments would be adversely affected. Consider two first-order streams which join to form a second-order stream. Assume the two watersheds drained by the first-order streams respond the same to snowmelt during rainfall so that their peak flows are additive at their confluence. If logging one watershed was to speed up snowmelt, its peak flow might occur earlier enough to desynchronize with the peak flow from the other watershed. Thus, peak flow below the confluence would be less than if both watersheds were forested. Increased runoff in the logged watershed could adversely affect the stream channel in that watershed, but below the confluence channel erosion could be less than if logging had not occurred.

That a generalization cannot be made about effects of timber harvest on snowmelt during rainfall, on size of peak flow, and on channel erosion can be illustrated by a second situation which is as plausible as the first. If flows from the two watersheds described above were not synchronized before logging, increased runoff caused by changes in snowmelt during rainfall might synchronize the flows--or it might desynchronize them further. Without understanding the rain-on-snow phenomenon and how it is affected by timber harvest, we have little chance of understanding how streamflow and erosion processes might be affected by timber harvest.

PREDICTING CHANGES IN STREAMFLOW

Whether or not the effects of timber harvest on the quantity and timing of streamflow in the Pacific Northwest can be predicted is of prime importance to forest land managers. This question is basic to the whole harvest scheduling idea and, of course, to this workshop. If we cannot reliably predict the consequences of our harvest activites on streamflow, then scheduling harvest activities according to some formal procedure designed to "maintain hydrologic balance" will be largely an academic excercise and probably will be ineffective in west-side Washington and Oregon.

A first step in formulating a procedure that could be used to help schedule timber harvesting for channel stability purposes is the

realization that channel erosion processes are dependent on the magnitude and duration of high flows. Implied in this step is the realization that there is no relationship between increases in annual water yield and increases in size of higher peak flows in western Washington and Oregon (Harr 1976a). It is true that increases in annual water yield after logging in experimental watersheds have generally been accompanied by higher peak flows in early fall and in spring and by relatively large increases in summer low flows. Neither change in streamflow, however, is involved in channel erosion processes.

Although the lack of relationship between annual yield and channel erosion in west-side Washington and Oregon has been well established, it has been either poorly communicated to land managers or not widely accepted by them. Consequently, annual water yield has been given much more than its fair share of importance and, as a result, may have been somewhat of a hindrance in understanding the effects of timber harvest on streamflow and channel erosion processes in this region. Possibly, the emphasis on annual water yield in both research and National Forest administration has stemmed from its emphasis in other regions, such as the Southwest (Barr 1956), the central Rocky Mountains (Goodell 1967), as well as the humid East (Hewlett and Hibbert 1961, Hewlett 1966, Hibbert 1967).

A number of computer models have been developed to simulate annual water yield. Even if these models could simulate yield changes, and few of them can, the relative accuracies of yield increases predicted by them or by the water yield analysis procedures of the type used in Region 1 (Galbraith 1973) would be irrelevant for west-side Washington and Oregon because of the aforementioned lack of linkage between water yield and channel erosion processes.

It should be apparent then that we must concentrate our efforts on predicting the effects of timber harvest activities on higher flows; for example, those above about 4 liters/sec.ha in figure 6. These flows are shaping channels in headwater basins in the Pacific Northwest. Also, synchronization of these levels of storm runoff from subwatersheds is most important in size and duration of high flows in higher order streams of parent watersheds. Once the processes controlling storm runoff in headwater basins are understood, we will stand a better chance of accurately predicting the effects of harvesting on the streamflow characteristics directly involved in channel erosion processes.

Unfortunately, we are not yet at that level of understanding. For a variety of reasons, studies of experimental watersheds have not included enough plot-level studies of processes conducted concurrently with watershed-level studies, so we still have only crude ideas about how a watershed produces streamflow and about what has caused observed changes in storm runoff after logging. We have circumstantial evidence linking increased size of peak flows to soil compaction which suggests, albeit roughly, that perhaps we should limit soil compaction—of the type found on haul roads, primary skidroads, and landings—to less than 10% of

total watershed area. But the answer is not that simple, for compaction on less than 10% of total watershed area could also increase size of peak flows if it were located on critical runoff-producing areas. And interception of subsurface water by roadcuts and ditches most likely is involved too. Logically, this could also route water to stream channels faster than natural subsurface flow processes do; thus, a given set of storm conditions that would produce a maximum flow insufficient to cause appreciable channel erosion in an undisturbed watershed could produce after roadbuilding a higher maximum flow sufficient to erode the channel.

To complicate the picture even more, there is the potential of increasing rate of snowmelt during rainfall by clearcut logging. Data from experimental watersheds in western Oregon show the importance of snowmelt during rainfall on peak flows and the occurrence of landslides (see footnote 5). Although snowmelt indices developed by the U.S. Army Corps of Engineers (1956, 1960) suggest that widespread removal of forest vegetation by clearcutting could increase rate of water input to soil and amount of storm runoff, effects of clearcutting on snowmelt from shallow packs during rainfall have not been demonstrated.

If this discussion of our present capability of predicting effects of timber harvest on streamflow characteristics most involved in channel erosion processes sounds negative to you then I have made my point well. I do not believe we can predict changes in size or duration of these high flows at this time. This lack of predictive capability stems primarily from our lack of understanding of (1) subsurface movement of water, (2) snowmelt from shallow snowpacks during rainfall, and (3) how each of these is influenced by timber harvest activities. This lack of predictive capability, however, must not be interpreted to mean certain timber harvest activities are not capable of damaging soil and water resources. For example, we cannot say that clearcutting in the zone of transient snowpacks will not increase size of peak flows caused by snowmelt during rainfall with any more confidence than we can say that clearcutting will increase the size of these flows.

There are some indications that eventually we should be able to predict the effects of timber harvest activities on the channel-eroding higher flows but when and to what extent we will be able to do this are unknown. Recent computer hydrology models have been able to simulate stormflow reasonably well. Simulated volume of storm runoff and size of peak flows in a small watershed in West Virginia averaged 89% and 96% of their respective measured values (Troendle 1979). A model developed for small watersheds in the H. J. Andrews Experimental Forest simulated storm hydrographs and would have fit measured data much better if a more accurate snow accumulation-melt submodel were available (Overton and White 1978). The subsurface routing mechanisms were critical parts of both models. Conceivably, improvements in such models could provide a framework for predicting changes in storm runoff provided, of course, we know how harvest activities change processes of the hydrologic system

that are critical to storm runoff and can express these changes in such a way that they can be included in the simulation models. We must know where and when road cuts and ditches will intercept subsurface flow of water, under what conditions soil compaction will affect peak flows, and how timber harvest affects rate of snowmelt during rainfall. These, I believe, are crucial to the success of any attempt to schedule timber harvest to protect soil and water resources. We must also keep in mind that how and where an activity is carried out may be much more important than if or when it is carried out.

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Herbicides in Soil and Water

Logan A. Norris

I. Patterns of chemical use in forestry

II. Pesticide regulation in forestry

III. Benefit - Risk assessments

  A. Benefits

  B Risks

  C. Direct and indirect effects

IV. The behavior of chemicals in the environment

V. The entry and fate of herbicides in soil

  A. Initial levels

  B. Additions

  C. Persistence

  D. Movement

VI. The entry and fate of herbicides in water

  A. Direct application

  B. Drift

  C. Mobilization in ephemeral streams

  D. Overland flow

  E. Leaching

VII. Planning and conducting effective monitoring programs

VIII. Future research emphasis

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Entry and Fate of Pesticides in Air[1]

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The entry and fate of pesticides in air has been a topic of increasing interest and importance in recent years. Although this might imply that it is a new topic, it arose as a problem after the Second World War in the earliest days of modern organic pesticide use. Problems of pesticide drift or volatilization from improper herbicide use were first indicated by plant damage and impacts on domestic animals (and in some cases even humans).

In the mid 1960s, growers of some crops which were quite sensitive to 2,4-D in the Milton-Freewater area of NE Oregon felt they suffered crop damage from widespread use of 2,4-D in nearby wheat culture areas. Scientists from Oregon State University found 2,4-D in some air samples, resulting in the enactment of regulations concerning the use of these materials. Similarly in Washington State, there are numerous regulations on the use of 2,4-D near areas where sensitive crops (like grapes) are grown.

Some damage has even been noted in forestry. In the late 1950s, several articles were published about "box elder blight," a disease that was occurring in some midwestern hardwoods. Later investigation showed that the cause was really 2,4-D, or similar chemical types, in the air. Other evidence of pesticides in air comes from reports of a wide range of pesticides present in rainwater. The U.S. Environmental Protection Agency is considering hearings on the "spray drift problem." These issues indicate that pesticides in air is an important topic, and one responsible applicators and planners should note.

SOURCES AND FORMS OF PESTICIDES IN AIR

Pesticides in air can be either solid, liquid or gas. Pesticides can be in the air in solid form when all the carrier has evaporated from a spray droplet, leaving only the pesticide as a solid behind. In other cases the pesticide may have been applied in a solid form, like granules or dust. However, we most often think of pesticides in the air as liquid and in the form of spray drift. Vapors can also be important and, although they are less apparent because we can't see them, we do see their effects. Volatilization is the process in which chemicals, in either the solid or liquid form, change into a gas. In forestry, of course, we are most often concerned with pesticides in the liquid and vapor states.

Where do pesticides in air come from? There are several sources. Manufacturing and waste disposal processes are one source and, if they were the only or most important source, we would have quite a different regulatory process. We know from experience, however, that pesticides enter the air during application. Since application is done at widely scattered locations, entry to air from this source becomes a much more difficult problem, both for applicators and regulators.

ENTRY TO AIR

Those of you involved with application have observed the entry of pesticides into air—one type of evidence is that cloud of spray which doesn't want to settle and appears to move from the target area. You may have observed the direct effects of spray damage from offsite deposition—hopefully not to desirable plant species on your neighbor's property.

Pesticides enter air in two general forms: *fine droplets* (spray drift) which do not fall in the target area and *vapor* from the volatilization of liquids. Unfortunately, most studies of the problem involve *indirect measurements*, that is, measurements of what gets to the area, rather than what is lost. No distinction is made between loss via drift or volatilization. Any chemical applied which does not reach the target can leave the area as a liquid in spray drift or as a vapor after volatilization.

Onsite Spray Deposition

There are some direct measures of spray deposition that have been used to estimate the amount of spray material which does not reach the target and is, therefore, assumed to be lost to the air. In foresry, early measurements by Newton, Norris, and Zavitkovzki showed that only 25 to 40 percent of a spray mixture applied by fixed-wing aircraft reached the ground in the Oregon Coast Range. These results imply substantial quantities of spray material did no reach the intended target. In our study, spray interception cards were placed on the ground, in "openings" in the brush but we had not allowed for the screening effect of the nearby vegetation. Research reported by Maksymiuk showed nearby vegetation can substantially reduce spray deposition at ground level (Table 1).

Table 1. Influence of nearby vegetation on spray deposit at ground level.

Horizontal Distance from Nearest Vegetation	*Spray Material Screened Out*
(tree heights)	*(percent)*
less than 1	43
1 to 2	16
2 to 3	9
more than 3	0

DDT applied in oil by fixed-wing aircraft from 750 feet. 200-micron droplets.

The data show that 43 percent of the spray material was screened out when the spray interception cards were less than one tree height away (horizontal distance) from nearby vegeta-

[1]This publicaton does not contain recommendations for pesticide use nor does it imply that the uses discussed here are registered. All pesticide uses must be registered by appropriate State and/or Federal agencies before recommendation.

tion. At distances equal to one to two tree heights, 16 percent interception occurred, while 9 percent interception was noted at distances greater than three tree heights. This particular data set is for DDT applied by fixed-wing aircraft flying at 750 feet above the canopy. It shows, however, the influence of nearby vegetation on spray drift measurements. Thus, what you measure on the ground is not always a good indication of the true rate of deposition in the target area.

In a later study conducted in connection with a helicopter application near Roseburg, a combination of 2,4-D and picloram was applied in a thin invert emulsion as part of a brush control/pasture renovation project. In this study 70 percent of the 2,4-D and 85 percent of the picloram were recovered at the top of the brush, the first intercepting layer between the aircraft and the ground. On powerline rights-of-way, I have recovered about 71 percent of the 2,4-D and 90 percent of the picloram at the first intercepting layer. These results would imply that from 10 to 30 percent of the spray material did not reach the "target." That may not be strictly true. The data say that *at least* that much spray reaches the target. If the capture efficiency of the spray cards is less than 100 percent, then the true rate of deposition will be greater than indicated by the cards.

Spray Droplet Capture Efficiency

Some data on spray droplet capture efficiency may help to illustrate this point. According to the data in Table 2, only 10 percent of 20-micron droplets traveling at one mile per hour on a collision course with a 1/8 inch rod (twig) will be captured (Figure 1). If the velocity is increased to 5 mph, capture efficiency increases to 50 percent. As the droplet size increases, capture efficiency also increases. At 10 mph, 10-micron droplets show a 30 percent capture rate compared to 65 percent for 20-micron droplets and 95 percent for 100-micron droplets (Table 3). Obviousy, the larger the droplet and the higher the velocity, the higher the capture efficiency.

Table 2. Influence of velocity on capture efficiency of 20-micron spray droplets by an 1/8 inch diameter twig.

Spray Drop Velocity	*CaptureEfficiency*
(miles per hour)	*(percent)*
1	10
5	50
20	75

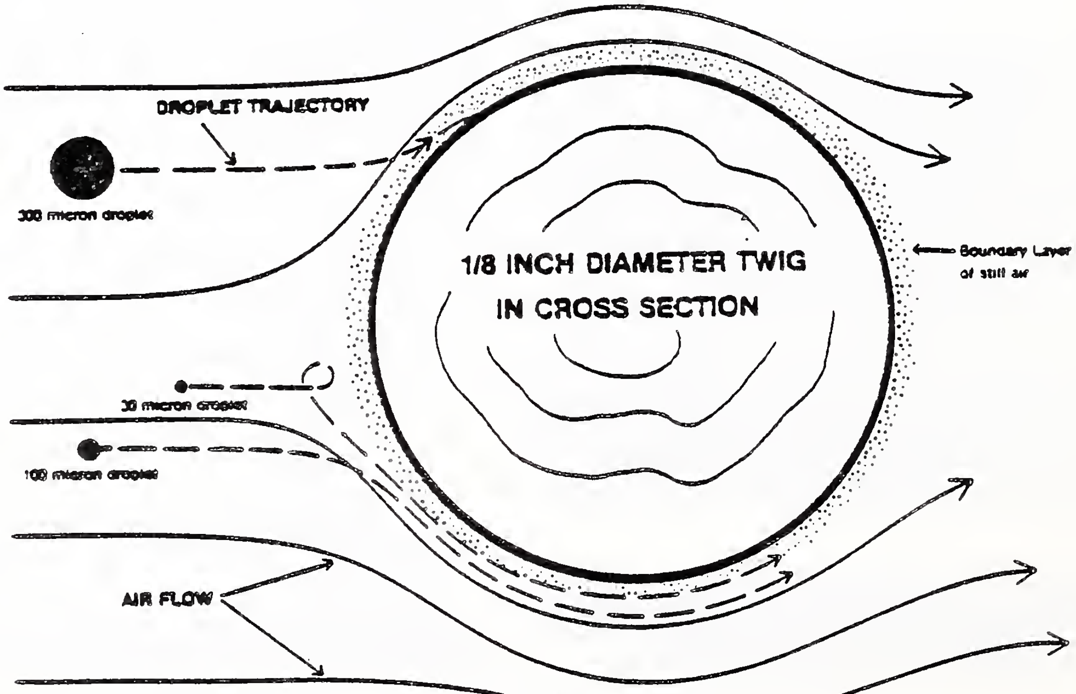
Table 3. Influence of droplet size on efficiency of capture by 1/8 inch diameter twig when spray droplet velocity is 10 miles per hour.

Droplet Size	*Capture Efficiency*
(microns)	*(percent)*
10	30
20	65
100	95

The reason these results are obtained is illustrated in Figure 1. As the stream of air, or the droplet moving through still air, reaches the intercepting surface, it tends to be deflected around the object. It is difficult for lightweight, slow-moving objects to penetrate the boundary layer of still air adjacent to the object. Clearly, the heavier (larger) the droplet and the faster its velocity, the more likely it will penetrate the boundary layer and be captured.

We have talked quite a bit about droplets of various sizes. What do these numbers mean in reference to things we are familiar with? A 50-micron droplet is quite small, about the smallest object you can see if you are looking at a single droplet. However, don't confuse this with looking at a spray clo[illegible] because obviously you can detect much smaller droplets a[illegible] group. A misty rain may have 500-micron droplets. The stand-

Figure 1. Deflection of small, slow-moving droplets around a 1/8 inch diameter twig.



ard dot on a printed page is about 1000-microns. This will give you some frame of reference for use in estimating droplet size.

'olatilization

The literature on volatilization has been summarized in three review articles which may help if you want to pursue the topic in greater detail. These references are:

W.F. Spencer, W.J. Farmer and M.M. Cliath. 1973. Residue Reviews Volume 49, page 1.

G.S. Hartley. 1967. Advances in Chemistry Volume 86, page 115.

G.A. Wheatley. 1976. Proceedings, British Crop Pest Control Symposium, page 117.

Gases, Liquids, and Vapor Pressure. Now let's talk a little more about gases, liquids, and the interrelations between these two states of matter. Figure 2 shows a layer of liquid in the bottom of a closed container. When this system has come to equilibrium, a proportion of the material which was originally in the liquid phase will evaporate and become a gas. As long as the temperature, pressure, and other conditions remain the same, the proportion of the chemical in the gas and the liquid phases will stay the same, that is, in equilibrium. If you attach a sensitive pressure gauge to the top of this container, it is possible to measure the pressure exerted by the vapor on the container walls. This illustrates what is meant by the term *vapor pressure*.

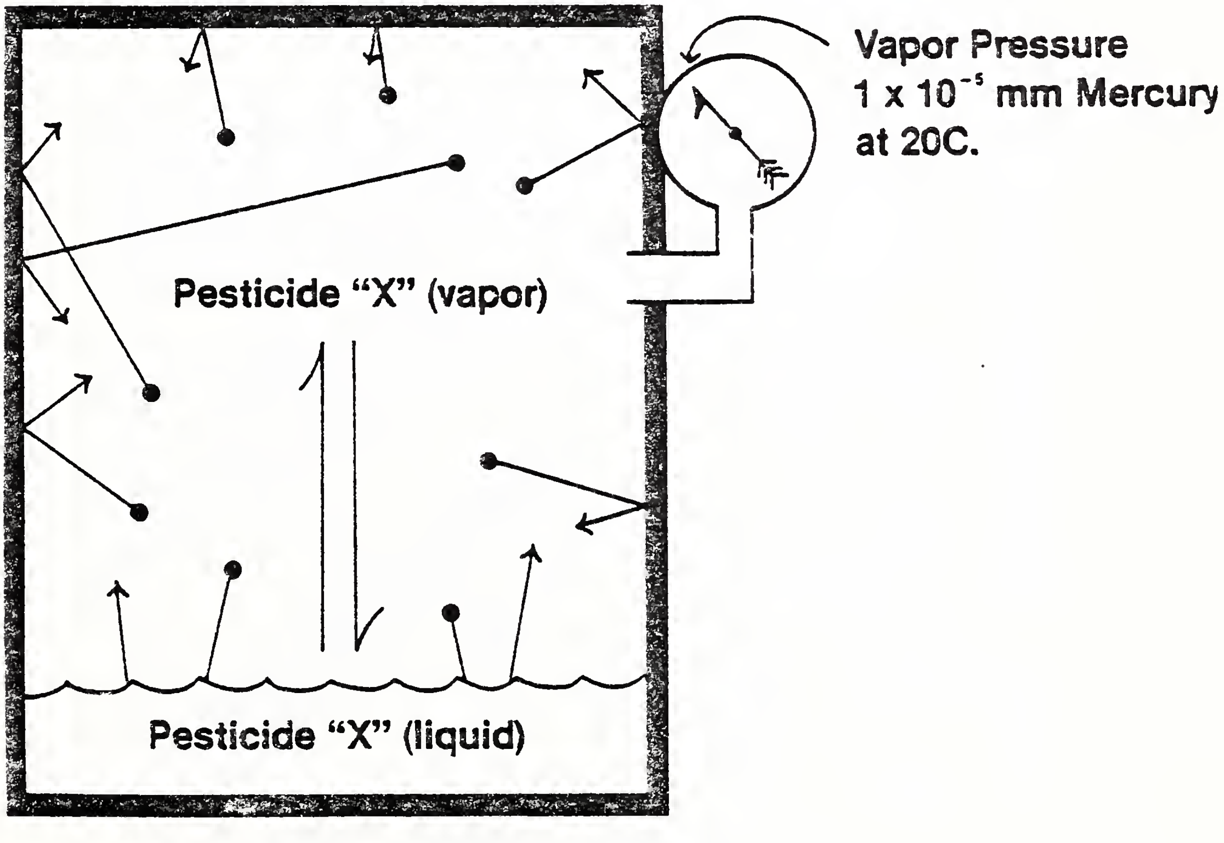
As an example, consider the high and low volatile esters of 2,4-D. High volatile esters tend to volatilize more readily and are more likely to cause offsite damage. A high volatile ester of 2,4-D would register a higher vapor pressure in the system shown in Figure 2 than would a low volatile ester. Thus we can see that chemicals with a high vapor pressure are more likely to volatilize than those with a low vapor pressure.

Factors Affecting Volatilization. There are several very practical factors that will influence the vaporization of any given chemical. You will encounter these factors in the field and should be aware of them and how they operate:

Heat. Volatilization occurs when the molecules in the liquid phase gain enough energy to increase their speed to the velocity needed to overcome the liquid's surface tension and escape into the air. The higher the temperature, the greater the energy level of the molecules, and the more likely they are to escape the liquid surface. Thus, increasing temperatures in the field will increase the rate of volatilization.

Surface Area. The greater the surface area to volume ratio, the greater the rate of volatilization will be, primarily because the distance any molecule will have to diffuse to reach the surface will be reduced. In the practical sense, the rate of evaporation will be higher from spray deposits which are very thin, than from large liquid droplets.

Figure 2. An illustration of the physical meaning of *vapor pressure.* In this closed container the amount of chemical X in the liquid phase is in equilibrium with the amount of chemical X in the vapor phase. At equilibrium, the pressure exerted by the molecules in the vapor phase on the walls of the container is the vapor pressure.



Dissipation Rate. This may also be called the *ventilation rate.* The faster molecules which have just entered the gas phase are moved away from the liquid surface, the faster the rate of volatilization. In the practical sense, as the velocity of the wind increases, the rate of volatilization will also increase.

Quantity Present. The larger the total amount of material present, the larger the amount of material that can evaporate. In the practical sense, the higher the rate of application, the greater the total amount of material that can evaporate.

Volatilization from Droplets. Volatilization occurs while a drop is falling through the air. The proportion of material which is recovered in droplets after falling through the air is proportional to the vapor pressure of the chemical involved. The higher the vapor pressure, the less the amount of material that will be recovered at the first intercepting surface.

Droplet size is important in determining both the length of time it takes a droplet to fall 50 feet and the length of time a droplet lasts before it is completely evaporated. At 86°F and 50 percent relative humidity, a 50-micron water droplet lasts only 3.5 seconds, not nearly long enough to fall 50 feet (3.4 minutes are required for a 50-micron droplet of constant size to fall 50 feet). A 200-micron droplet does better, and will last for 56 seconds, only requiring 13 seconds to fall 50 feet.

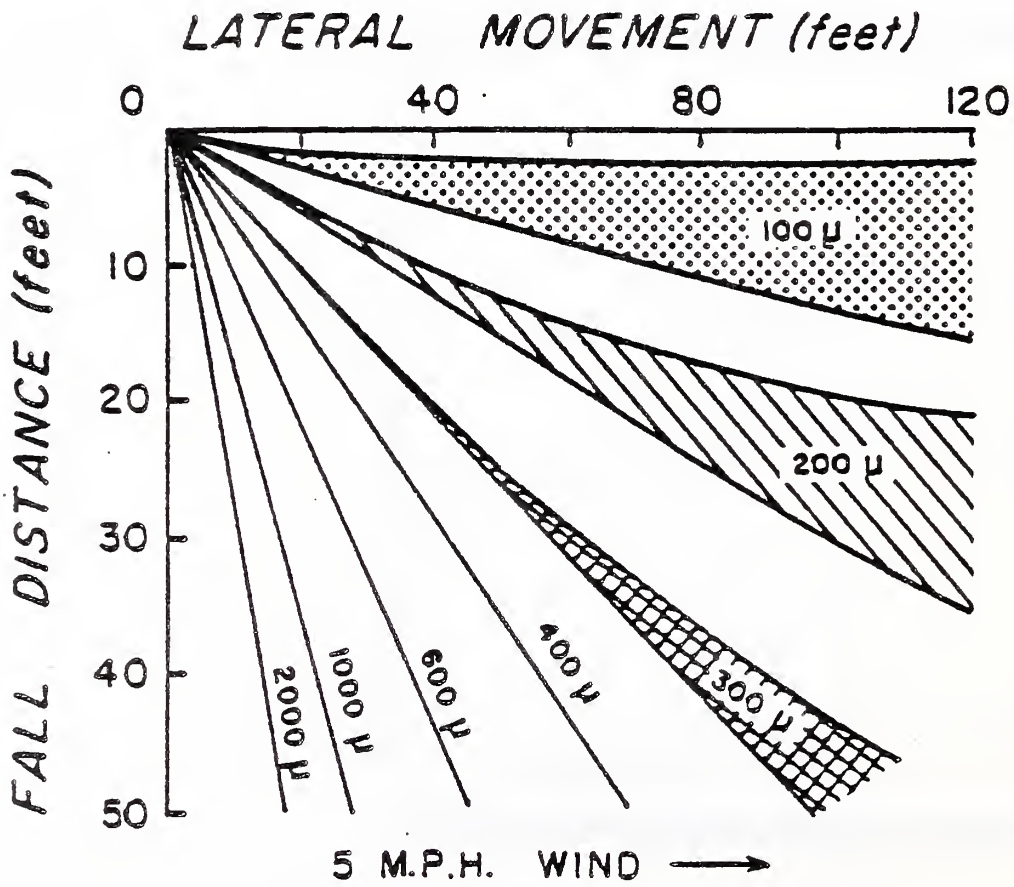
The effect of initial droplet size, wind velocity, and height of release on droplet lateral movement (in a 5 mph wind) as they fall through the air is often illustrated through the use of data, as in Figure 3. The assumption is that droplets remain the same size as they fall, but they do not. Both the carrier and the pesticid- evaporate and the droplet gets smaller. Figure 4 incorporates th- concept of evaporation into the type of data presented in Fi- 3. In this case, the liquid involved is water. Obviously -or compounds with a lower vapor pressure than water, or at a lower temperature and higher relative humidity, the effect would be reduced, i.e., the rate of evaporation would be less and the droplets would not get smaller as rapidly.

According to Figure 4, an 80-micron water droplet will fall only two feet and drift laterally about seven feet before it has completely evaporated! A 150-micron droplet will fall about 15 feet and drift about 22 feet before it has completely evaporated.

In the examples above, the droplet disappeared. What happened to the chemicals which made up the droplet? They did not disappear, and only changed from a liquid to a gas. In the gas phase, they still retain their biological activity and, if present in sufficient quantity, can cause damage to offsite vegetation and animals.

Volatilization from surfaces. Droplets are not the only chemical source which evaporate and enter the air. Once a spray droplet has fallen onto a surface, like a leaf, the evaporation process can continue. The nature of the surface, however, makes a lot of difference. Liquid evaporation will be most rapid and complete from a truly inert surface. The data in Table 4 contrasts the evaporation of 2,4-D butyl ester from a pyrex slide

Figure 3. The influence of droplet size and height of release on lateral movement of spray droplets in a five mile per hour wind. The shaded area indicates uncertainty due to droplet evaporation.



and a leaf. Obviously evaporation was greater from the inert glass surface than from the leaf. Similar results would be obtained if evaporation from a glass surface were contrasted with loss from a soil surface. The interaction between the chemical and the surface it contacts will greatly influence the rate of evaporation. The greater the degree of interaction, the slower and less extensive the evaporation of the pesticide. This is analogous to the leaching of chemicals in the soil. The greater the degree of interaction between the chemical and the soil, the less leaching will occur. Many of the same physical-chemical principles are involved.

Table 4. Distribution of 2,4-D butyl ester after application to a leaf and a pyrex glass surface.

	Pyrex glass	*Leaf*
	(percent found)	*(percent found)*
Vapor phase	99	41
On the surface	0.2	2
Inside	0	57

Losses from surfaces can be appreciable for some pesticides. At 86°F, losses equal to 4.5 lb/A per year of DDT, 19.6 lb/A per year of dieldrin, and 178 lb/A per year of lindane have been measured in the laboratory.

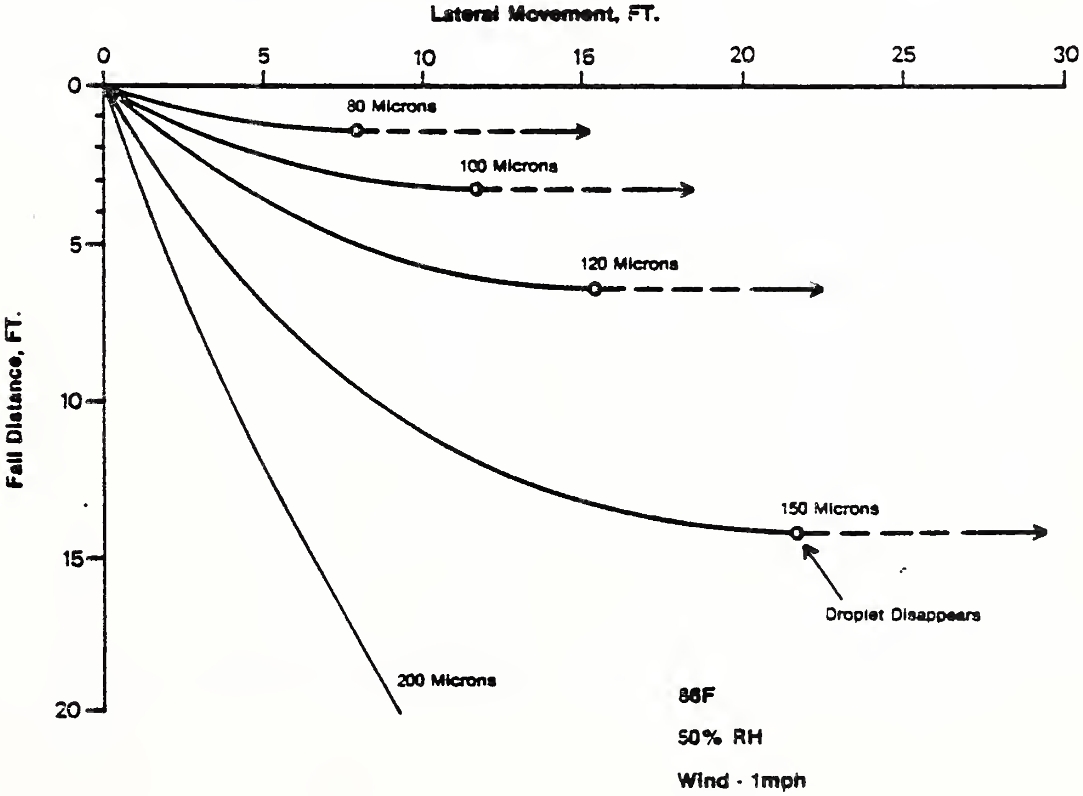
FATE IN AIR

Several things can happen to a pesticide once it has evaporated. Dispersal by the wind and dilution by the large mass of uncontaminated air is common. Pesticides in the vapor phase can also (a) be adsorbed onto surfaces of all kinds—like vegetation, soil and even skin, (b) be washed out with falling rain or other forms of precipitation, (c) dissolve in liquids—like surface water, or (d) undergo photodecomposition. Unfortunately, there is little quantitative data on decomposition rates for most pesticides in the natural environment. The laboratory studies which have been conducted illustrate that photodecomposition is possible and many products of degradation have been identified, but not under field conditions.

CONCLUSIONS

I hope this information has given you a better appreciation for the entry and fate of pesticides in air. Clearly, drift and volatility are important factors in determining the actual or potential risks involved with the use of specific pesticides. Unfortunately, there is little quantitative data relative to field conditions to guide us in this area. Future research programs should emphasize this topic to help obtain a data base as extensive as the one we have for chemicals in water and soil. These data bases have served well in both administrative and regulatory efforts to insure the safe use of forest chemicals. A similar data base for pesticides in air is needed for the same purpose.

Figure 4. The influence of initial droplet size on vertical and horizontal movement of water droplets in one mile per hour wind, 86°F, and 50 percent relative humidity. In this figure, droplet size decreases with time because of evaporation. Open circles indicate the point at which the droplet has completely evaporated.



Behavior of Chemicals in the Forest Environment[1]

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CHEMICALS USED IN FORESTRY

Chemicals play an important role in agriculture and forestry. The levels of chemicals that are being used in the United States for various functions in forestry are shown in Table 1.

The use of these materials in agriculture is very large compared to their use in forestry. It seems out of persective that forestry, which represents less than 1 percent of the use, generates most of the complaints. The level of use of various pesticides by the USDA Forest Service is in Table 2.

Most of the insecticide applications that are made in forestry in the United States involve Federal programs in some way and are, therefore, included in this compilation. Thus the data on insecticides probably represents the majority of their use in forestry in the United States. The data for herbicides is only for use on National Forest land and is therefore a substantial under-estimate of the total use of herbicides in forestry. Keeping those points in mind, it is clear that the level of use of herbicides far exceeds the level of use of insecticides in forestry in the United States.

Tables 3 and 4 list the most commonly used herbicides and insecticides. Although the list looks long, in fact the vast majority of use is accounted for in a relatively few compounds. For instance 2,4-D and picloram, either alone or in combination

[1]This paper was presented at "Forest Pesticides Shortcourse," sponsored by Oregon State and Washington State Universities, Portland, Oregon, March 11, 1981. This publication does not contain recommendations for pesticide use nor does it imply that the uses discussed here have been registered. All pesticide uses must be registered by appropriate State and/or Federal agencies before recommendation.

Table 1. Comparative annual use of chemicals in agriculture and forestry.

Pesticides (1976)	Agriculture (10^3 kilogrms)	USDA Forest Service (10^3 kilograms)
Insecticides	73,682	68
Herbicides	179,227	183
Fungicides	19,636	5
Fertilizers (1978)	Agriculture (10^3 metric tons)	Forestry (10^3 metric tons)
Nitrogen	9,636	55
Phosphorus	2,273	5
Fire Retardants		
Year	Quantity Used (liters)	User Group
1955	87,000	all users
1961	28,400,000	all users
1966	22,500,000	USDA
1966	12,200,000	Calif.Div.For.
1966	3,800,000	BLM
1970	64,400,000	all users
1977	55,000,000	USDA

Table 2. Pesticides used by USDA Forest Service

Pesticide	October 1, 1978 to September 30, 1979[1]			October 1, 1979 to September 30, 1980[2]		
	Acres[3]	Pounds[4]	Percent of Total	Acres[3]	Pounds[4]	Percent of Total
Herbicides	184,047	471,184	63	229,877	371,748	62
Insecticides	272,420	173,000	23	173,500	156,590	26
Fumigants	297	70,636	9	216	42,419	7
Fungicides	1,037	10,629	1	2,159	20,596	3
Repellents	9,502	9,137	1	8,998	2,723	
Rodenticides	44,920	9,066	1	29,693	2,063	
Wood preservatives				——	1,564	
Piscicides	239	915		256	633	
Algacides	54	408		17	429	
Behavorial chemicals	2,270	17				
TOTALS	514,786	744,992		444,716	598,765	

[1] Pesticide-use advisory memorandum 246 (2140 pesticide-use management, June 5, 1980), USDA Forest Service, Washington, DC.
[2] Pesticide-use advisory memorandum 284 (2150 pesticide-use management and coordination, March 12, 1981), USDA Forest Service, Washington, DC.
[3] One acre = 0.4 ha.
[4] One pound = 0.45 kg.

Table 3. Herbicides used by USDA Forest Service

Herbicide	October 1, 1978 to September 30, 1979[1] Acres[3]	Pounds[4]	Percent of Total	October 1, 1979 to September 30, 1980[2] Acres[3]	Pounds[4]	Percent of Total
2,4—D	73,449	185,323	39.3	117,174	205,676	55.3
Picloram	15,853	24,947	5.3	68,001	35,849	9.6
Atrazine	5,299	18,915	4.0	7,099	29,772	8.0
Mineral spirits	89	6,600	1.4	50	15,615	4.2
2,4-D + Picloram	57,00	135,307	28.7	9,090	12,670	3.4
Glyphosate	3,668	5,841	1.2	5,667	9,601	2.6
Dalapon	4,240	10,611	2.3	4,076	9,084	2.4
Fosamine ammonium	1,949	7,939	1.7	1,649	8,340	2.2
2,4-D + 2,4-DP	3,152	8,947	1.9	1,777	6,877	1.8
Simazine	4,298	9,927	2.1	1,016	4,790	1.3
Dicamba	1,060	1,404	0.3	3,714	4,207	1.1
Hexazinone	383	839	0.2	2,768	3,900	1.0
Sodium metaborate + sodium chlorate	11	793	0.2	11	3,700	1.0
Ammonium Sulfamate	450	3,502	0.7	173	1,983	0.5
Amitrole	880	1,710	0.4	822	1,902	0.5
MSMA	3,558	18,604	3.9	1,386	421	0.4
2,4-D + Dicamba	6,232	14,971	3.2	288	164	

[1] Pesticide-use advisory memorandum 246 (2140 pesticide-use management, June 5, 1980), USDA Forest Service, Washington, DC.
[2] Pesticide-use advisory memorandum 284 (2150 pesticide-use management and coordination, March 12, 1981), USDA Forest Service, Washington, DC.
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Table 4. Pesticides used by USDA Forest Service

Pesticide	October 1, 1978 to September 30, 1979[1] Acres[3]	Pounds[4]	Percent of Total	October 1, 1979 to September 30, 1980[2] Acres[3]	Pounds[4]	Percent of Total
Malathion	193,364	93,512	54.1	141,485	102,347	65.3
Carbaryl	51,178	50,508	29.2	26,091	29,687	19.0
Azinphosmethyl		4,324	2.5		17,923	11.4
Carbofuran		5,469	3.2		4,223	2.7
Toxaphene		2,295	1.3		1,607	1.0
Lindane	370	309	0.2		443	
Diazinon	101	124		64	81	
Tetraclorvinphos	31	68		58	54	
Ethylene dibromide		2,521	1.5		40	
Acephate	23,400	11,706	6.7		3	

[1] Pesticide-use advisory memorandum 246 (2140 pesticide-use management, June 5, 1980), USDA Forest Service, Washington, DC.
[2] Pesticide-use advisory memorandum 284 (2150 pesticide-use management and coordination, March 12, 1981), USDA Forest Service, Washington, DC.
[3] One acre = 0.4 ha.
[4] One pound = 0.45 kg.

with other herbicides, accounted for 70 percent of herbicide use in fiscal year 1980. A similar pattern is observed for insecticides. The level of use of particular herbicides holds fairly constant from year to year. The use of particular insecticides varies much more on a year-to-year basis.

INDIRECT VS. DIRECT EFFECTS

The use of a pesticide can produce both direct and indirect effects (Figure 1). A direct effect requires that there be some direct physical contact between the organism and the chemical. If this does not take place, there can be no direct chemical effect as we have defined it.

Figure 1. The contrast between direct and indirect effects.

DIRECT AND INDIRECT EFFECTS

DIRECT EFFECTS - REQUIRES DIRECT PHYSICAL CONTACT BETWEEN THE ORGANISM AND THE CHEMICAL.
EXAMPLE: 2,4-D SPRAYED ON ALDER CAUSES THE ALDER TO DIE.
2,4 D DRIFT DEPOSITED ON TOMATOES CAUSES REDUCED FRUIT YIELD.
INDIRECT EFFECTS - THOSE EFFECTS WHICH DO NOT RESULT FROM THE DIRECT PHYSICAL CONTACT BETWEEN THE CHEMICAL AND THE ORGANISM.
EXAMPLE: USE OF 2,4-D REDUCES THE AVAILABILITY OF FORBS FOR GOPHERS SO THEY LEAVE.

An example of a direct chemical effect is: 2,4-D sprayed on alder causes the alder to die. This is a direct chemical effect on a target organism. Or: 2,4-D drift deposited on tomatoes causes reduced fruit yield. This is also a direct effect, but on a non-target organism. The mehanisms and the requirements for direct effects are the same for both target and non-target organisms.

Indirect effects do not require a direct physical contact between the chemical and the organism. For many forest chemicals, the predominant environmental effect is indirect. This is certainly true with herbicides. Herbicides cause changes in the composition and density of the vegetation which in turn produce some indirect effects. For example, 2,4-D use reduces the availability of forbes, causing gophers to leave. This is an example of an indirect effect.

Much of our decision making regarding the use of chemicals concentrates on the direct effect. Substantially less attention is given to indirect effects.

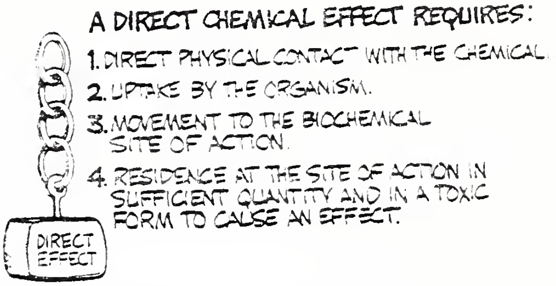
There are several steps necessary before a direct chemical effect can occur (Figure 2). It is like an event chain. If the chain breaks at any one point, the subsequent step cannot take place. For instance, if the chemical and the organism come in direct physical contact but there is no chemical uptake by the organism, the subsequent steps cannot take place, preventing manifestation of the final direct effect.

RISK ASSESSMENT

Any reasonable assessment of hazard or risk has two components. One is the consideration of chemical toxicity. The second is evaluation of the potential for exposure. If both components are not included, you are as likely to reach the wrong conclusion as the right one.

Toxicity is the component we hear most about. The use of chemicals which are very highly toxic may be low in risk if non-target organisms do not receive substantial exposure. On the other hand, use of a compound that is very low in toxicity

Figure 2. The steps which must take place in order for a direct chemical effect to occur.



could present substantial risk if organisms receive a continuing and substantial exposure. The point is, both of these elements, toxicity and exposure, are important. They both must be considered in making an assessment of risk.

MARGIN OF SAFETY

One way of expressing the degree of risk or hazard is through the margin of safety. The margin of safety is the ratio of the no-effect level to the exposure level (Figure 3). A margin of safety of one, or less, means the exposure level is equal to or greater than the no-effect level, and you may have a problem. If the margin of safety is greater than one, the level of exposure is less than the no-effect level. The lower the level of exposure is relative to the no-effect level, the larger the margin of safety.

Figure 3. An illustration of a margin of safety.

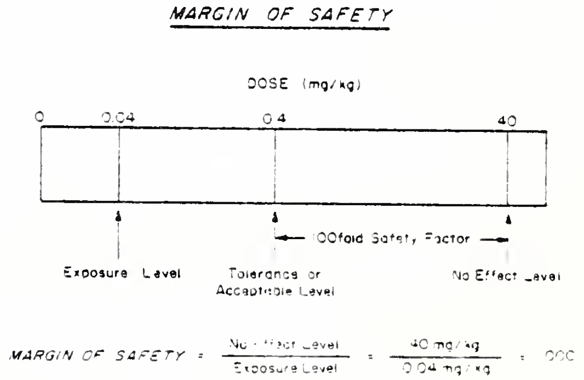


Figure 3 is a hypothetical example. In this case, the no-effect level is 40 mg/kg and the exposure level is 0.04mg/kg, to give a margin of safety of 1,000. That is, the level of exposure is 1,000 times less than the level required to produce the first effect (or just the no-effect level). A margin of safety of 1,000 seems quite good, however, all margins of safety considered as acceptable are not the same. Society and regulatory agencies apply different standards to margins of safety for different materials. For instance, margins of safety for some common medications are small, surprisingly small, numbers, like 1.5 to 15. Margins of safety of 100 have been commonly applied for pesticide tolerances in food, some pharmaceuticals and food additives. Margins of safety of at least 1,000 appear to have been required by EPA for 2,4,5-T.

TOXICITY AND EXPOSURE

There are two general kinds of toxicity, acute and chronic

(Figure 4). Acute toxicity is the type of response we associate with exposure to a few, relatively large doses, which are delivered over a short period of time. Chronic toxicity is associated with exposure to many, relatively small doses, delivered over a relatively long period of time. The point is that the nature of the response is determined by the nature of the exposure. If the behavior of a chemical in the environment is such that chronic exposure does not occur, chronic effects will not occur. The nature of exposure is dependent on the behavior of the chemical in the environment.

Figure 4. The difference between acute and chronic toxicity—exposure.

THERE ARE TWO KINDS OF TOXICITY

ACUTE – Few, large doses, short term exposure

CHRONIC – Many, small doses, long term exposure

BEHAVIOR IN THE ENVIRONMENT

The behavior of a chemical in the environment includes its movement, persistence and fate (Figure 5). The properties of chemicals (melting points, vapor pressures, water solubilities, etc.) are important factors. In addition, the environment also has its own properties; things like incoming radiation, the presence and movement of moisture and surfaces of all kinds. The properties of chemicals interact with the properties of the environment in a way that is guided by the laws of nature. The result is the movement, persistence and fate of chemicals we observe in the environment (Figure 6). These determine the potential for exposure because the behavior determines how much chemical is in the environment, for what period of time and in what form.

Figure 5. The behavior of a chemical includes—.

BEHAVIOR OF A CHEMICAL

is its

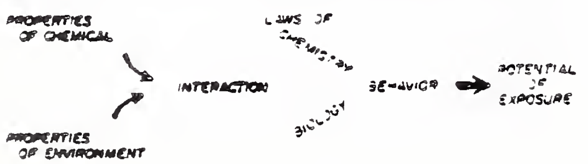
MOVEMENT

PERSISTENCE &

FATE

in the environment

Figure 6. The properties of a chemical interact with the properties of the environment to produce the behavior in the environment we observe and the potential for exposure.

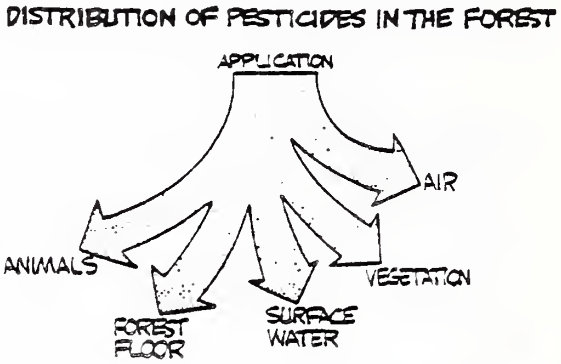


There are scientific reasons why a particular chemical behaves in the environment the way it does. This means we can study, understand and, as we get smarter, learn to predict with increasing accuracy the behavior of chemicals in the environment. This will improve the likelihood that we will make the right estimate of potentials for exposure and, therefore, better estimates of the true margin of safety.

INITIAL DISTRIBUTION

What really happens when a chemical is applied in the forest environment? It is distributed initially among five compartments (Figure 7). The size of the arrows are not quantitative.

Figure 7. Aerially applied pesticides are distributed initially to five compartments of the forest environment.



How much chemical ends up in any given compartment is affected by such a wide variety of things it is beyond the scope of this paper. Based on my experience and research, the initial (and maximum) levels in some compartments are listed in Table 5. There is relatively little published data on residue levels in air and animals. As research continues, better estimates for these compartments will be available.

Table 5. Maximum initial concentrations of pesticides in the forest.

Vegetation	200 ppm per pound per acre
Forest Floor or Soil	0.5 ppm per pound per acre
Water	0.368 ppm per acre foot per pound per acre (The more usual range is 0 to 0.02 ppm.)

MOVEMENT, PERSISTENCE AND FATE

Once the chemical is distributed among the five compartments of the forest, several processes begin to affect the chemical. All of them can be grouped into three main processes: transport, storage and degradation. Temporary storage occurs when a chemical is adsorbed onto a surface. Unless something happens to that chemical while it is stored, it will eventually be released back into the environment.

Transport functions include volatilization, leaching and surface runoff and are important in defining the level of exposure in any particular location. For the most part, transport only moves material from one part of the environment to another, resulting in a reduction in concentration at one location and an increase in concentration at another. It really is only

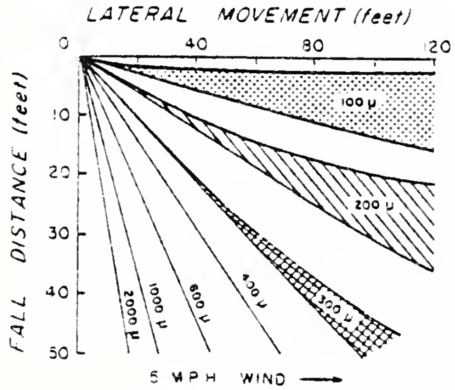
through degradation that permanent removal of the chemical from the environment is achieved.

Air

From a limited amount of data, we know that a certain amount of pesticide is dispersed in the air. Unfortunately, the air is the compartment we know least about.

Droplet size is important in determining the degree of drift (Figure 8). Big droplets fall faster and move less laterally than do small droplets. Applicators and planners have a fair degree of control over what happens in the air simply by influencing the size of the droplet. The type of equipment and formulations used, atmospheric conditions during time of application and the operating conditions of equipment are important factors in determining droplet size.

Figure 8. The influence of wind speed, droplet size and fall distance on lateral movement of spray droplets (drift).



However, do not get the impression that big droplets are good and small droplets are bad. You can go too far by making great big drops and, while you do not get any drift, you do not achieve any brush or insect control either. Use care in considering this topic. Apply existing technology to minimize drift in those situations where it is really necessary to minimize drift.

Vegetation

What happens to chemicals that fall onto vegetation? Table 6 illustrates the concentration of three different herbicides on grass at different times after application. Note that the highest levels occur on vegetation immediately after application. The level of chemical on and in that level of vegetation decreases as a function of time. Volatilization, photodecomposition, plant metabolism and weathering processes are all processes that operate to reduce the level of chemical on vegetation as a function of time after the application.

Table 6. Residues of Herbicide[1] in Forage Grass.

Time after Treatment	Herbicide Residue 2,4-D	2,4,5-T	Picloram
(weeks)	(ppm)	(ppm)	(ppm)
0	100	100	135
1	60	60	
2	50	30	32
4	30	15	
8	6	6	24
16	1	2	16
52	—	—	3

[1]Rate of application was 1.12 kg/ha.

What does 25 ppm on the vegetation mean relative to the well-being of rabbits or deer that might be consuming this particular vegetation. By combining information from Table 6 with information on toxicity, you can assess the likelihood of significant risk or hazard in connection with a particular herbicide use.

Forest Floor and Soil

Much of the chemical that is applied in the forest ultimately ends up in the forest floor or soil. Some falls directly on the forest floor during application. In addition, material deposited on vegetation is transported to the forest floor in falling leaves or in through-fall precipitation.

Adsorption is a particularly important process that occurs in the forest floor and soil. It is an equilibrium phenomenon (Figure 9). Some chemical remains in the soil solution and some is physically and chemically bound to soil minerals or organic matter. These amounts remain proportionally the same, i.e., in equilibrium. If some of the chemical in the soil solution is degraded by micro-organisms or volatilized back into the atmosphere, the chemical in the bound form is released to the soil solution to reestablish the equilibrium. Adsorption has implications for persistence and leaching of chemicals in the soil profile.

Figure 9. The adsorption equilibrium equation.

$$\text{CHEMICAL} + \text{ADSORBENT} \underset{k_2}{\overset{k_1}{\rightleftharpoons}} \text{CHEMICAL ADSORBENT}$$

Figure 10 shows the relative mobility of a number of chemicals, some of which are important in forestry. The chlorinated hydrocarbon insecticides are the least mobile. Some of the herbicides used are among the most mobile on this relative scale. This is only a relative scale, however, and even compounds that are relatively mobile, like picloram, actually move only short distances. In the forest, almost without exception, the vast majority (over 90 percent) of the chemical is found in the top six inches of soil. There is relatively little chemical deeper than 18 inches.

Figure 10. The relative mobility of a number of common pesticides in soil.

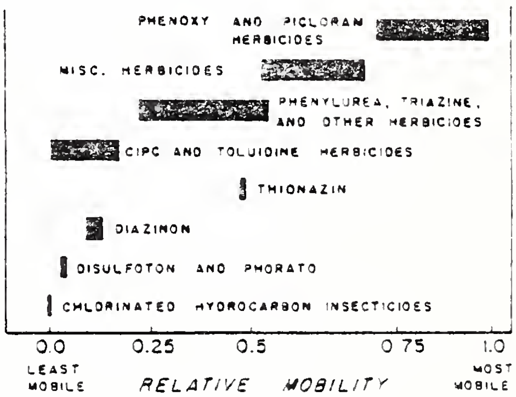


Figure 11. The persistence of a number of common pesticides in soil.

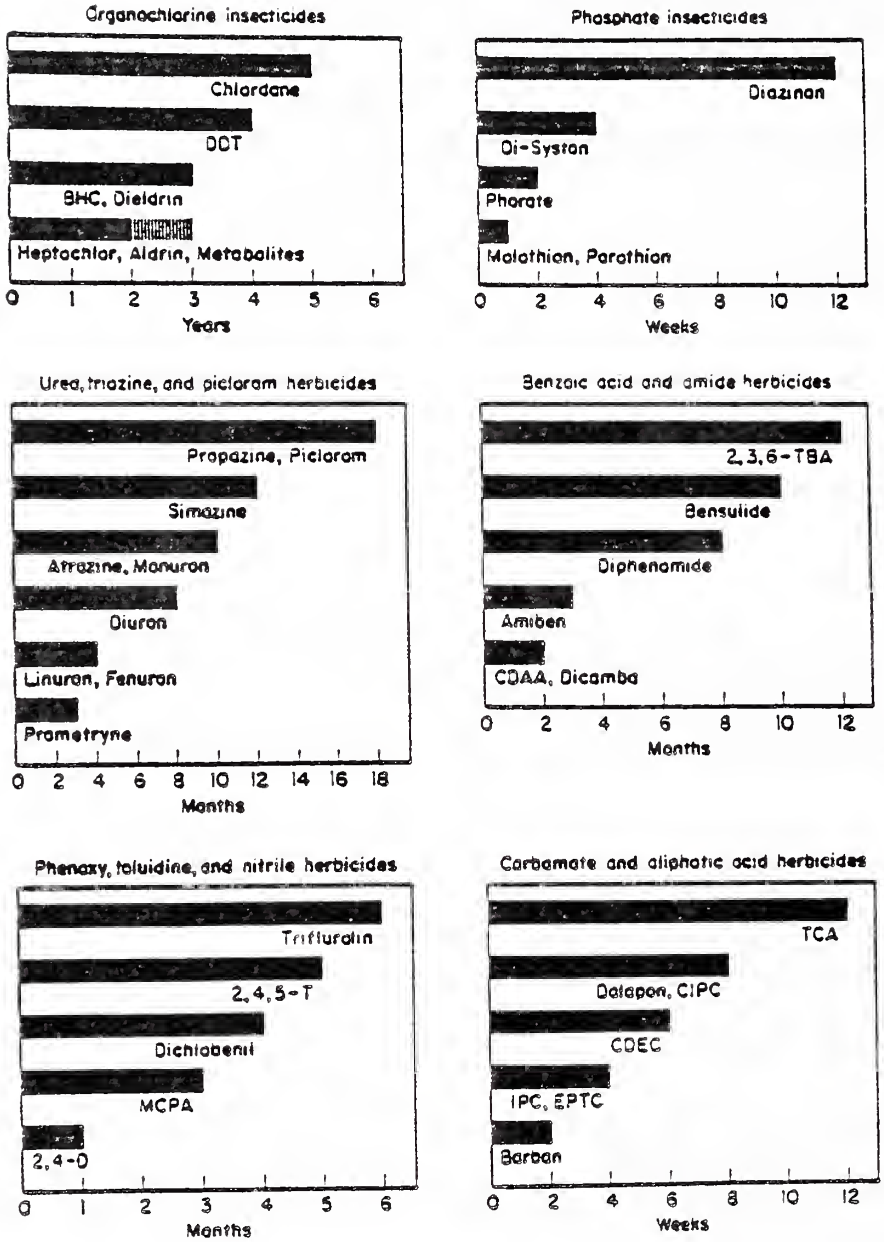


Figure 11 shows data on the persistence for a large number of pesticides, some of which are used in forestry. Note the different time scales. Figure 12 illustrates the persistence of four herbicides in the forest floor. All of these chemicals disappear, but not at the same rate. Even materials we think are highly persistent are subject to decomposition. For instance, we think of picloram as a relatively persistent material, and certainly it is persistent relative to 2,4-D and amitrole. However, the level of picloram does decrease with time. As time goes on, less and less of these chemicals will be in the forest floor and soil environment.

Animals

Chemicals are in animals because one or both of two phenomena occur (Figure 13). Bioaccumulation is the uptake and temporary storage of a chemical from the environment by an organism. This type of exposure occurs when a fish is exposed to a chemical in water. The chemical is taken into the fish and will be present in the flesh at some level higher than in the water. Biomagnification results from the movement of chemical through the various tropic levels (up the food chain). There is relatively little data on either bioaccumulation or biomagnification from field studies for forest pesticides.

The importance of bioaccumulation and biomagnification varies greatly with the chemical, the organism and the nature of the exposure. As a general rule, compounds which are highly water soluble and have only limited fat solubility tend to be much less susceptible to either bioaccumulation or biomagnification. These are characteristics of most chemicals used in forestry. This does not mean that animals captured in spray areas within a day of application would not contain residues. It means that residues may be found at low levels, but only for the short period of time the organism is exposed. As exposure decreases, the chemical is rapidly excreted. Chemicals that are highly fat soluble and low in water solubility may reside in the body for a longer period of time.

Figure 13. The difference between bioaccumulation and biomagnification.

BIOACCUMULATION - THE UPTAKE AND TEMPORARY STORAGE OF A CHEMICAL FROM THE ENVIRONMENT BY AN ORGANISM.
EXAMPLE: FISH EXPOSED TO 1 PPT DDT IN WATER MAY CONTAIN 10,000 PPT IN THEIR FLESH.

BIOMAGNIFICATION - THE INCREASE IN CONCENTRATION OF A CHEMICAL AS IT MOVES FROM A LOWER TO A HIGHER TROPHIC LEVEL.
EXAMPLE: BIRDS WHICH EAT FISH CONTAINING 1 PPM MAY HAVE 100 PPM IN THEIR FLESH.

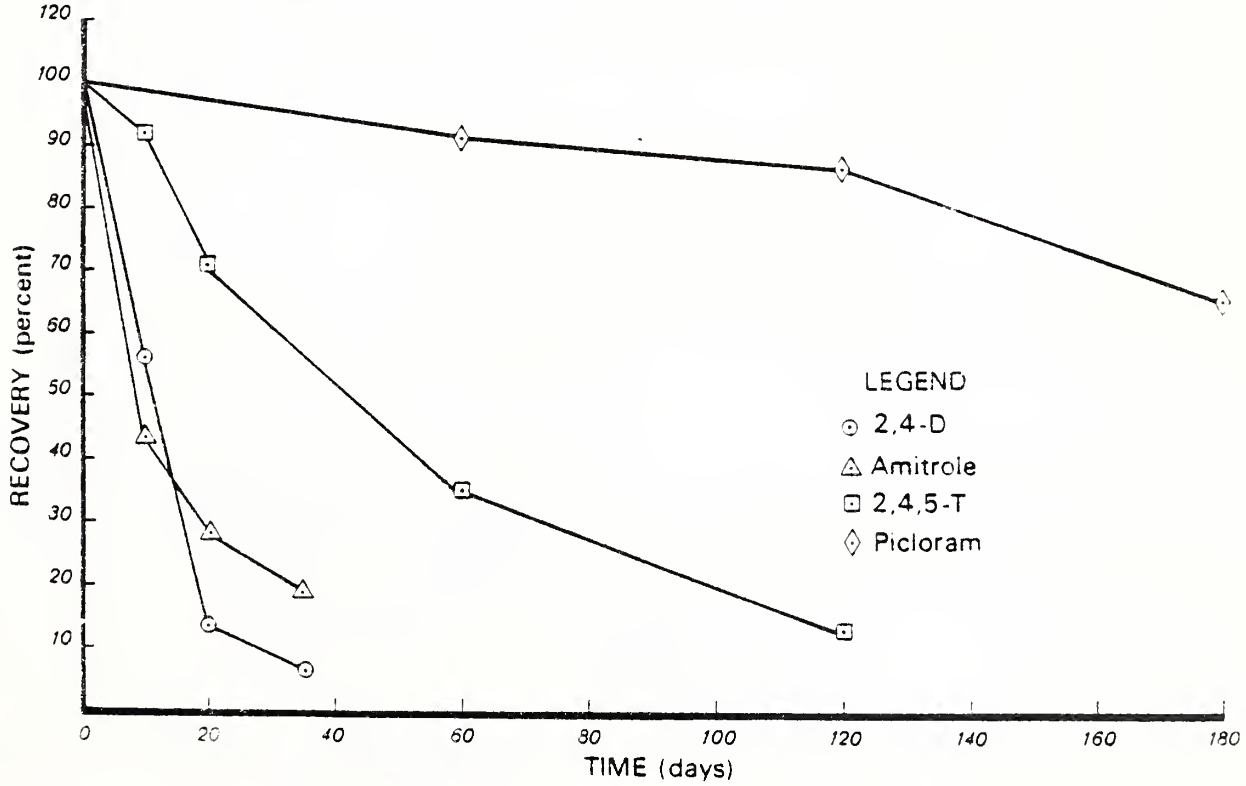
Water

Water is an important component of the forest and the one we know most about. Chemicals enter forest streams by several mechanisms (Table 7). Each mechanism has different conse-

Table 7. Characteristics of herbicide entry to streams.

Mechanism of entry	*Relative size of peak concentration*	*Duration of entry*
Drift and Direct application	High	Short—only during application
Mobilization in Ephemeral stream channels	Moderate to low	Short—only during first storms after application
Overland flow	Low	Medium
Leaching	Very low	Long

Figure 12. The persistence of four herbicides in the forest floor.



quences in terms of magnitude and duration of exposure of stream organisms or downstream water users. Direct application and drift has the potential for the highest peak concentrations, but the duration of entry will be short.

Mobilization of chemicals in ephemeral (intermittent) stream channels is also an important process in chemical entry to water. Chemicals deposited in a dry stream channel are subject to mobilization by any water moving through that channel after the application. The concentration will usually be low and the duration of entry will be short because it will usually occur only during those first few storms after application.

Let us look at some data on water contamination. The area in Figure 14a in western Oregon was sprayed with 2,4,5-T. Note the locations of sprayed areas, streams and water sampling sites. Except near Sample Point 1, a one-swath buffer strip was left between the treated area and the stream.

Figure 14a. The Cascade Creek Watershed.

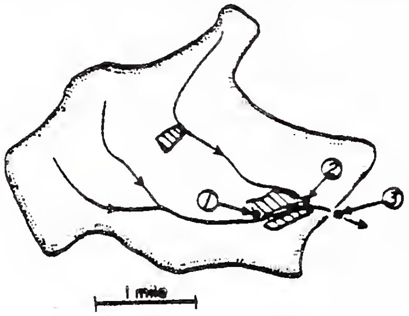
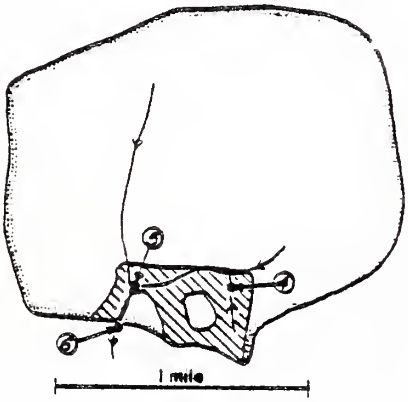


Figure 14b. The Eddyville Watershed.



Water samples from Sample Points 2 and 3 contained highest concentrations shortly after the application. Then the levels decreased rapidly, so that after four or five hours residues were no longer detected (Table 8). Sampling continued at this point for over one year but no further residues were found. From data like this, we believe a principle entry mechanism into streams is direct application and drift to the surface water.

At a second location in the same general area, the stream either originated within or flowed directly through the treatment unit. There was no buffer strip. Obviously, direct application to the stream surface occurred (Figure 14b). Highest residues occurred shortly after application and then decreased rapidly with time (Table 9). Drift and direct application onto surface water was the most important mechanism of entry.

Table 8. Concentration of 2,4,5-T in Cascade Creek[1].

Sample Point 1		*Sample Point 2*		*Sample Point 3*	
Hours after spraying	*2,4,5-T (ppb)*	*Hours after spraying*	*2,4,5-T (ppb)*	*Hours after spraying*	*2,4,5-T (ppb)*
0.05	0	0.2	1	0.3	Lost
0.6	16	1.3	2	1.4	3
1.3	7	2.2	1	2.0	3
2.0	4	3.9	1	3.9[3]	0
4.0	4	5.4[3]	0		
5.2	4				
9.8	4				
24.7	2				
48.2	1				
74.8[2]	1				

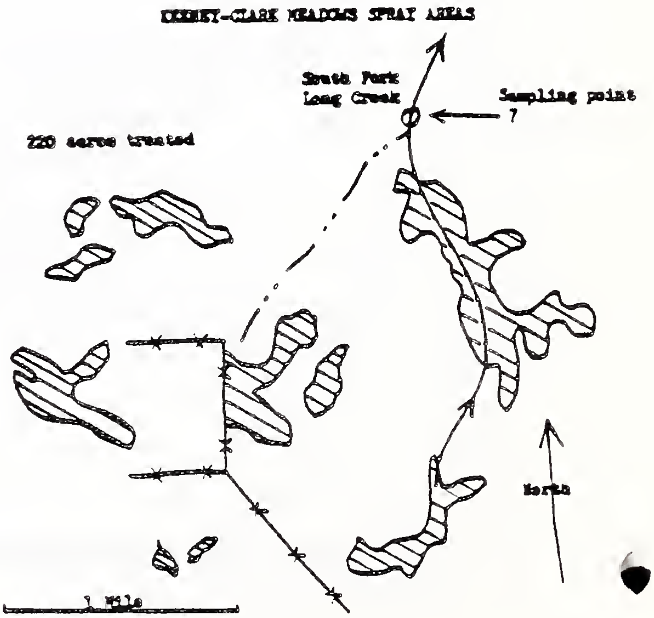
[1]Rate of application = 2.24 kg/ha.
[2]Herbicide residues were detected at Point 1 up to 16 weeks after spraying.
[3]No further residues were detected, althoug sampling continued for 10 months.

Table 9. Concentration of 2,4-D in streams near Eddyville.[1]

Sample Point 4		*Sample Point 5*		*Sample Point 6*	
Hours after spraying	*2,4-D (ppb)*	*Hours after spraying*	*2,4-D (ppb)*	*Hours after spraying*	*2,4-D (ppb)*
0.8	33	1.	62	1.4	30
1.8	13	2.3	71	2.3	44
2.8	13	3.3	58	3.3	25
53.5[3]	9	4.3	44	4.3	23
		53.6[2]	25	53.6[2]	11

[1]Rate of application = 2.5 to 3.36 kg/ha.
[2]No further residues were detected although sampling continued for 10 months.

Wet marshy areas are a peculiar type of situation you need to be aware of. The area in Figure 15 was treated with 2,4-D but no



buffer strips were used along streams. In addition, this meadow area was very wet. The highest concentrations of herbicide occurred shortly after the application (Table 10). These levels are higher than observed most other places and the residues were detected for long periods of time after the application. We have learned that this is the pattern expected from making herbicide applications to marshy, wet areas that have high water tables. These areas are usually characterized by wet-land plants and can be identified in advance and excluded from the treatent area to avoid contamination. In situations like this, a relatively slight rise in the water level due to precipitation flushes relatively large quantities of chemicals into the stream system. In terms of water contamination, I view this as the one that has the greatest potential for causing dangerous levels of stream contamination, yet is controllable because you can identify these areas and exclude them from treatment.

Table 10. Concentration of 2,4-D in streams in Keeney-Clark Meadows.

Hours after spraying	*2,4-D*
0.7	840
2.5	48
3.1	128
3.6	106
4.1	106
6.1	121
8.1	176
9.6	138
14.3	113
37.8	91
56.4	76
100.1	115
103.6	95
289.9	5
297.0	7

[1]Rate of application = 2.24 kg/ha.

The area in Figure 16 was sprayed with dicamba, which is somewhat persistent and highly mobile in the soil. If long-term entry of chemicals into the stream occurs, we expect to see it with this compound. Sample Point 1 was closest to the treatment unit. The highest levels occurred shortly after the application and then decreased rapidly with time (Figure 17). After 37 hours, residues were no longer detected, even though sampling continued for 1 year.

At downstream locations, the concentrations tend to be lower and the time at which peaks occur tend to be displaced with time as you move further and further downstream. There was some loss of chemical from the water as it moved between

Figure 16. The Farmer Creek spray area.

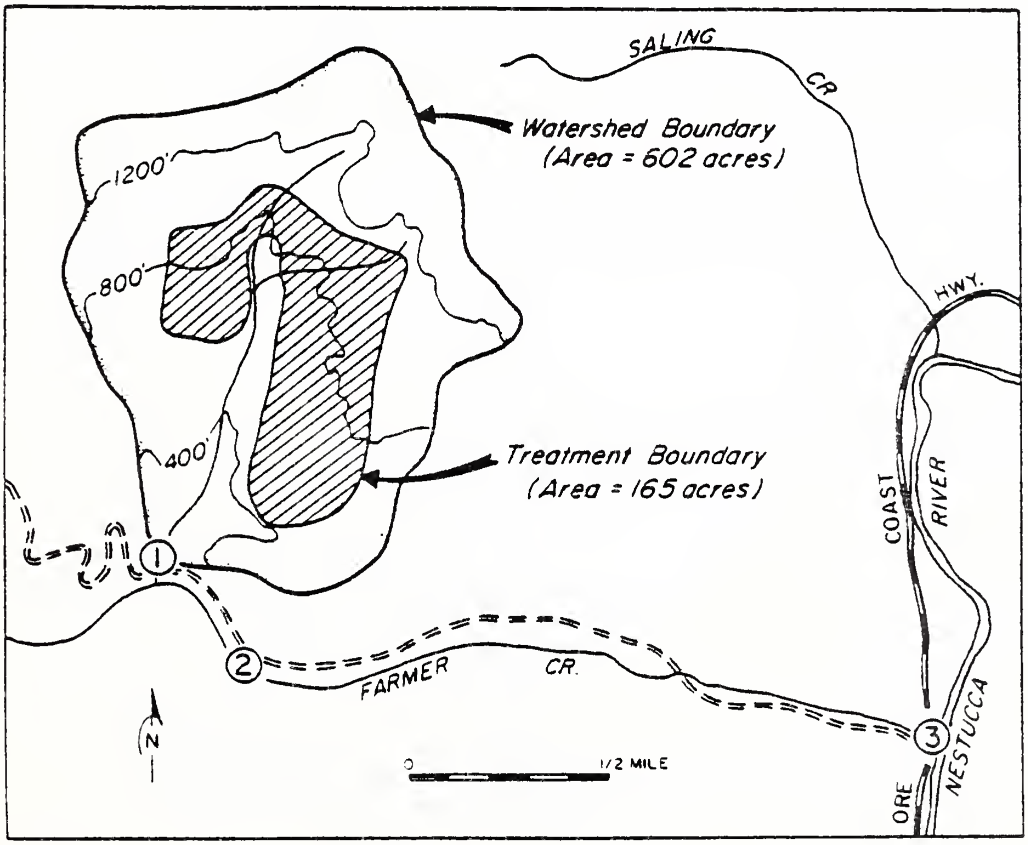
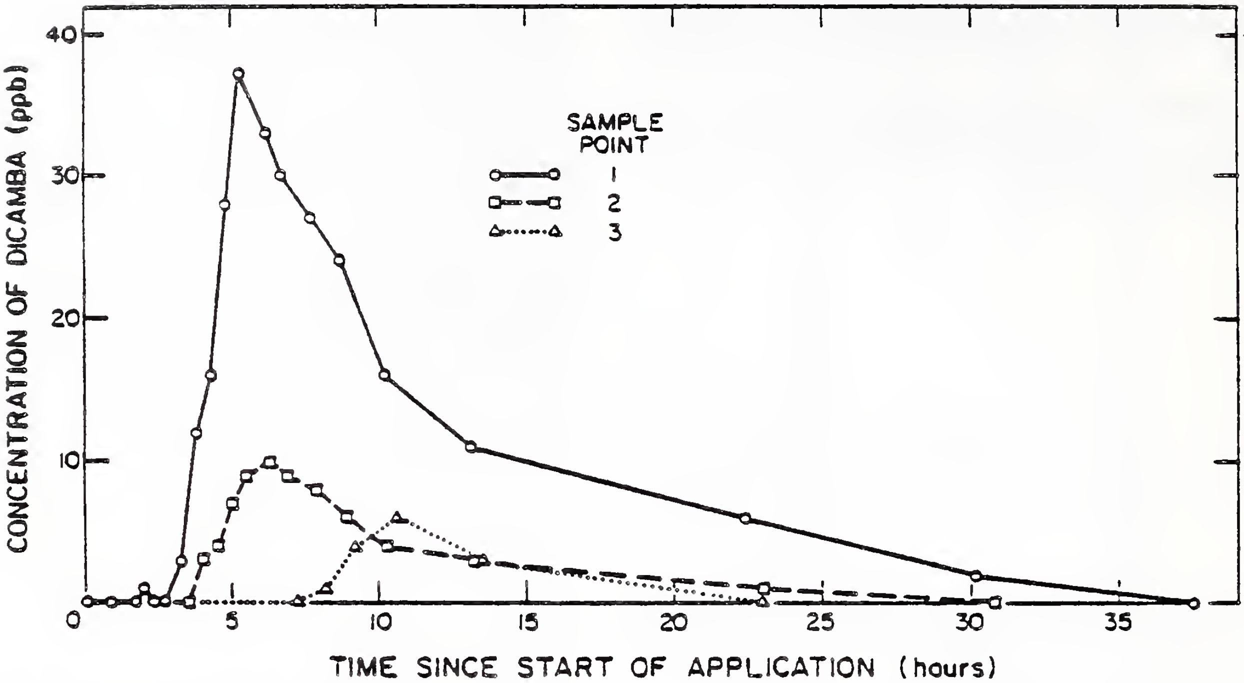


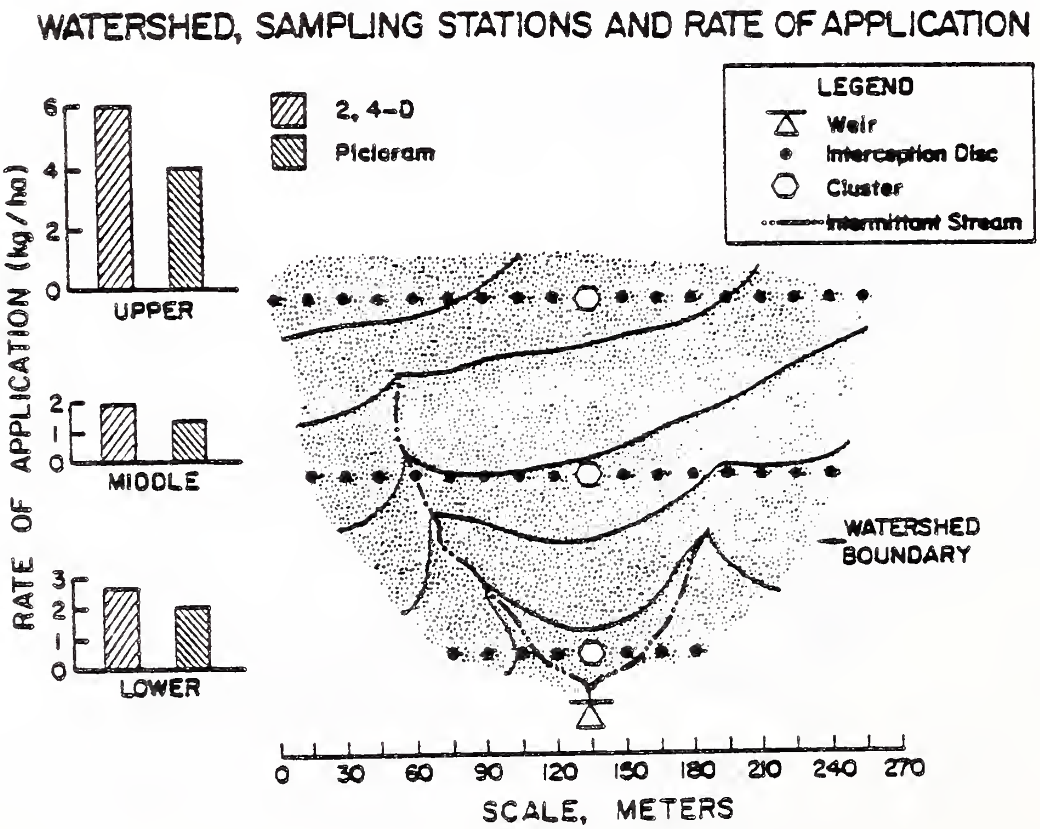
Figure 17. The concentration of dicamba in streamwater at several points in the Farmer Creek spray area.



Points 2 and 3. Dilution, volatilization, adsorption on sediments and uptake by organisms are the reasons the residue patterns change as the herbicide moves downstream.

Mobilization in ephemeral stream channels is also an important topic. The area in Figure 18, about 16 acres, was sprayed with 2,4-D and picloram. The intermittent streams did not carry

Figure 18. The Roseburg spray area.



water at the time of the application. Note the data in Figure 19 comparing precipitation, stream water discharge (water discharged from the watershed at any particular point in time), and herbicide discharge (or grams of herbicide discharged from that watershed). There was little rain at this site until September. The first rain was not sufficient to cause stream water discharge. Later, as soil recharge was accomplished, more rain caused both water and chemical discharge. Most of the chemical was discharged during the storms of late December. Even though the concentration was quite low, the large volume of water discharged resulted in a greater discharge of herbicide. Through the course of the study, 0.02 percent of the 2,4-D and 0.35 percent of the picloram were discharged in stream water. The stream channel occupied 0.2 percent of the total watershed area. Mobilization in these ephemeral stream channels was the most important process in transporting chemicals from this watershed.

We can sum our knowledge of water contamination in one statement: If you do not want it in the water, don't put it there. There is no magic involved. Those of you involved in planning and applying herbicides can eliminate or minimize the entry of chemicals into water by identifying those areas where water is present. Avoid direct application or drift in those areas and you will reduce or eliminate the levels of chemical that may occur in water.

PROPERTIES OF CHEMICALS

Many of the important chemical, physical and biological properties of chemicals used in forestry are in Table 11.

2,4-D

The amine salts and mineral salts of 2,4-D are quite soluble in water. This is important in determining how they are going to behave in the soil and the potential for bioaccumulation. Esters, on the other hand, tend to be much lower in water solubility and they may be volatile. Esters hydrolize rapidly to acid or salt forms. 2,4-D exhibits relatively little leaching and does not persist for long periods in the soil (half-life of one to four weeks). The half-life in vegetation is a week or less. There is little potential for bioaccumulation.

Picloram

Picloram is fairly water soluble. It leaches more than 2,4-D but less than dicamba and is moderately persistent in soil (most will be gone in one year). It exhibits little or no bioaccumulation.

Atrazine

Atrazine is low in waer solubility and low in volatility (more volatile than probably either picloram or 2,4-D when not present in the ester form). It adsorbs on soil much more strongly than either 2,4-D or picloram, but less than simazine. Atrazine is relatively short in persistence in the soil (80 percent is gone in 90 days). It is rapidly metabolized in plants, particularly those that are resistant to it and has little potential for bioaccumulation.

Glyphosate

Glyphosate is highly water soluble (10,000 ppm), has negligible volatility, is strongly adsorbed on soil, and does not leach. Its half-life in soil is 30 to 60 days with 90 percent of it gone in 12

Figure 19. Precipitation, streamwater discharge and herbicide discharge at the Roseburg study area.

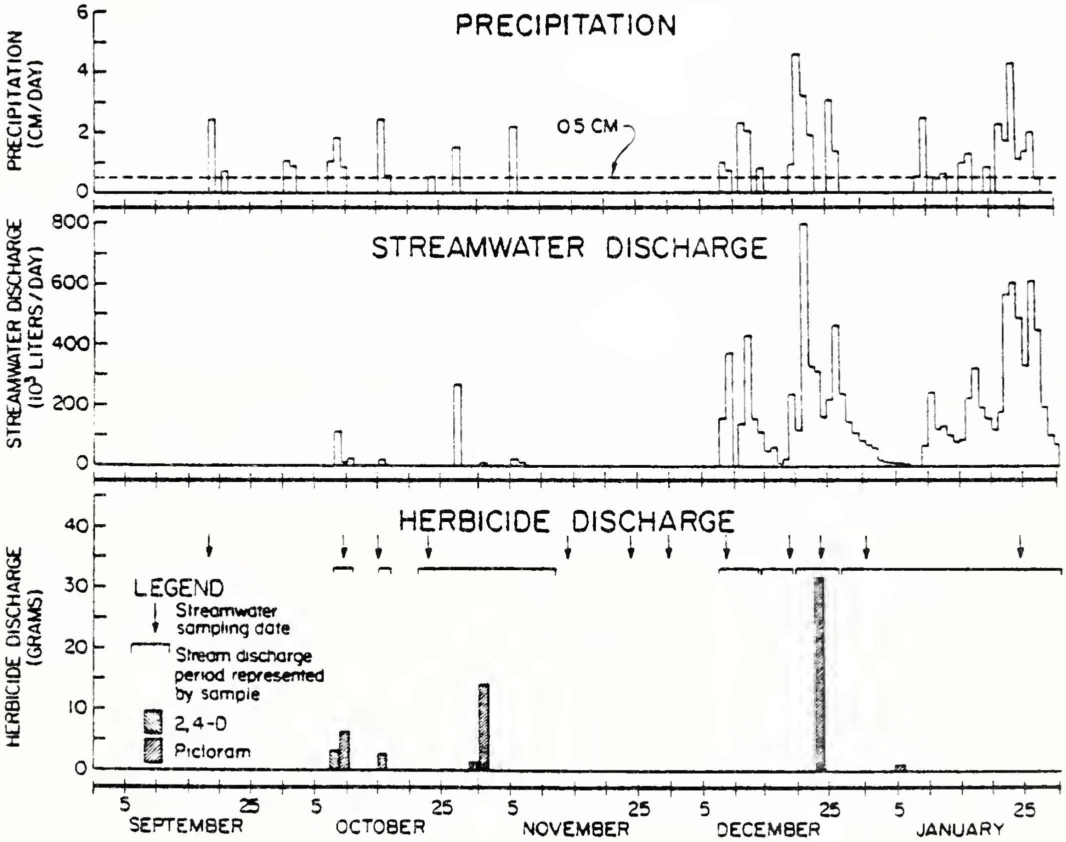


Table 11. Properties, behavior in the environment and toxicity of selected chemicals.

Chemical	*Chemical Properties*	*Behavior in the Environment*	*Toxicity*
Atrazine	Low water solubility, 30 ppm; low volatility	Adsorbs in soil but less so than simazine; life in soil: 80% gone in 90 days; rapidly metabolized in plants; little potential for bioaccumulation	LD_{50} 1750-3080 mg/kg 100 ppm 2 years—rats no effect LC_{50} 4-100 ppm fish
Dicamba	High solubility in water, 7900 ppm; low volatility	Low adsorption to soil; leaches readily (more than picloram); half-life in soil—14 days; no bioaccumulation	LD_{50} 1000-3000 mg/kg 500 ppm 2 years—dog, rat no effect LC_{50} 20 ppm bluegill
Fosamine Ammonium (Krenite)	Highly water soluble, 179 g in 100 g water; low volatility	Rapidly deactivated in soil; half-life in soil: 1 week; no appreciable leaching; half-life in vegetation: 2-3 weeks; no bioaccumulation	LD_{50} 700-2400 mg/kg 1000 ppm 90 days—rats LD_{50} 670 ppm bluegills, 1000 ppm—trout
Glyphosate (Roundup)	Highly water soluble (10,000 ppm), negligible volatility	Rapidly inactivated in soil, strongly adsorbed, no leaching; half-life in soil 30-60 days, persistent in plant tissues; no bioaccumulation	LD_{50} 4900 mg/kg 300 ppm 2 years—dog or rat, no effect LC_{50} 2-40 ppm—fish
Picloram	Water solubility, 430 ppm; low volatility	Some adsorption in soil; leaches more than 2,4-D, less than dicamba; moderately persistent in soil although most is gone in one year; no bioaccumulation	LD_{50} 2000-8000 mg/kg 150 mg/kg/day—2 years, dog no effect 1 ppm in water—10 weeks no effect on aquatics
2,4-D	Amine salts—very water soluble, 300 g/100g; esters—low water solubility, 50-150 ppm; esters may be volatile, but hydrolize rapidly	Adsorbs on soil, little leaching; half-life in soil, 1-4 weeks; little potential for bioaccumulation	LD_{50} 375-700 mg/kg 1250 ppm-lifetime, rats no effect Aquatics usually not affected at concentrations less than 0.1 ppm in water; drinking water standard standard 0.1 ppm

weeks. Although somewhat persistent in plant tissues, glyphosate will not bioaccumulate.

Fosamine Ammonium

Fosamine ammonium is highly water soluble (197 grams will dissolve in 100 grams of water), low in volatility and is rapidly deactivated in the soil (one-week half-life in soil). It does not leach appreciably. The half-life in vegetation is 2 to 3 weeks and there is no bioaccumulation.

CONCLUSION

All pesticides must be registered by the U.S. Environmental Protection Agency. This process requires that information of the type presented above be considered in the registration process. This means chemicals which are registered are reasonably effective and safe if used carefully in accordance with the label.

The movement, persistence, and fate of the phenoxy herbicides and TCDD in the forest

By

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Contents

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Residue Reviews, Volume 80

I. Introduction

The phenoxy herbicides 2,4-dichlorophenoxyacetic acid (2,4-D), 2,4,5-trichlorophenoxyacetic acid (2,4,5-T), 2-(2,4-dichlorophenoxy) propionic acid (dichlorprop), 2-(2,4,5-trichlorophenoxy)propionic acid (silvex), and 2-methyl-4-chlorophenoxyacetic acid (MCPA) are important tools for weed and brush control.[1] They have played important roles in the culture of agricultural crops, forest and rangeland management, and a wide variety of noncropland weed control programs.

There has been increasing controversy about their safety since the late 1960s when 2,4-D and 2,4,5-T were used in Vietnam. More recently, questions have also been raised about their utility in achieving vegetation management objectives. In about 1969, 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD), a highly toxic trace contaminant, was reported in trichlorophenol-based pesticides such as 2,4,5-T and silvex. Most recently, the U.S. Environmental Protection Agency (EPA) has initiated administrative hearings regarding the future registration of 2,4,5-T and silvex. 2,4-D is also being evaluated by EPA in a process which could lead to similar hearings on that compound. Presumably all the phenoxy herbicides may ultimately be involved. The heart of this regulatory process is the risk-benefit assessment. While there is some argument about benefits from using phenoxy herbicides (including the utility of alternative chemical, fire, mechanical, and hand-labor methods for controlling unwanted vegetation), the central issue in the controversy is safety (risk assessment). This issue has been raised and argued most intensely in connection with the use of these herbicides in forestry.

An adequate evaluation of risk requires information about (1) the toxicology of the chemical, particularly dose-response relationships for acute and chronic responses, and (2) the magnitude and duration of exposure organisms will receive. These characteristics of exposure result from the behavior (*i.e.*, movement, persistence, and fate) of the chemical in the environment (NORRIS 1971).

This review compiles the published data on the behavior of phenoxy herbicides and TCDD in the forest environment. Emphasis has been given to studies done in the forest or which utilize substrates from the forest. References to studies in the agricultural environment or the

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[1] Mention of pesticides and proprietary products does not constitute an endorsement or recommendation for use by USDA nor does it imply registration under FIFRA as amended.

laboratory have been included as necessary to illustrate important concepts, to reinforce forest-oriented studies, or to bridge critical gaps in knowledge.

The *National Research Council of Canada* (1978) compiled an extensive monograph on the phenoxy herbicides. It is an excellent source of data, but it emphasizes the agricultural environment. A Society of American Foresters workshop on 2,4,5-T in forestry in 1977 produced six summary papers which covered the 2,4,5-T controversy, environmental behavior, toxicology, economics, and methods for managing vegetation in forest and rangelands (NORRIS 1977 a and b, SCHWETZ 1977, HONING and ROSS 1977, WALSTAD 1977, NEWTON 1977, DALEN 1977). These papers, along with those by DOST (1978), TURNER (1977), and YOUNG *et al.* (1978) and the symposium proceedings on chlorinated phenoxy acids from the Royal Swedish National Academy of Science (*Swedish National Scientific Research Council* 1978), provide good recent summaries on the phenoxy herbicides, but their emphasis is on 2,4,5-T and their scope extends beyond the forest environment. KIMMINS (1975) reviewed the ecological effects of herbicide usage in forestry, including some data on the behavior of phenoxy herbicides in the forest. Extensive bibliographies of the phenoxy herbicides and the substituted dibenzo-*p*-dioxins are also available (DIAZ-COLON and BOVEY 1976, 1977, and 1978, BOVEY and DIAZ-COLON 1977, 1978 a and b).

II. Characteristics of the forest and the use of phenoxy herbicides

The forest environment is quite different from the agricultural environment; therefore, the behavior and impact of the phenoxy herbicides in the forest are also likely to be different. Forest land is typically at higher elevations, receives more precipitation, and has steeper sloping land which is usually more fully occupied by vegetation on a year-round basis than is agricultural land. Forest soils are typically shallow, have a very high infiltration capacity, low pH, and high organic matter content, and are overlain by forest floor material consisting of a litter (L) layer on top of a partially decomposed (F) layer which is on top of a humus (H) layer.

The phenoxy herbicides are used extensively in forestry for a variety of vegetation control purposes. 2,4-D is the most commonly used herbicide in forestry. 2,4,5-T received its most extensive use there. Silvex, dichlorprop, MCPA, and the other phenoxy compounds receive much less use in the forest. It is difficult to determine accurately the annual use of phenoxy herbicides in forestry in the United States. The USDA Forest Service compiles data on the use of herbicides on National Forest land, and these data can be used as a guide to the relative amount of use among the various phenoxy herbicides.

Data for two time periods are included in Table I because the amount

Table I. *Use of phenoxy herbicides on National Forests.*[a]

Herbicide	Fiscal year 1976[b]		Fiscal year 1978[c]	
	Acres	Pounds	Acres	Pounds
2,4-D	136,195	240,204	183,465	246,530
2,4,5-T	53,251	102,720	4,161	6,841
dichlorprop	6,073	8,068	13,921	16,127
silvex	1,198	3,755	238	365

[a] Land administered by the USDA Forest Service as part of the National Forest System. 1 lb = 0.454 kg, 1 A = 0.405 ha.
[b] Derived from Pesticide Use Advisory Memorandum 167, April 14, 1967. USDA Forest Service, Washington, D.C. Covers 15 months, July 1, 1975 through September 30, 1976.
[c] Derived from Pesticide Use Advisory Memorandum 219, May 18, 1979. USDA Forest Service, Washington, D.C. Covers 12 months, October 1, 1977 through September 30, 1978.

of use has changed as public pressure and administrative, regulatory, and judicial processes have influenced decision-makers. The use of 2,4,5-T in particular has been affected. Table II shows the level of use of this herbicide from fiscal years 1971 through 1978.

Phenoxy herbicides are applied to the forest in a variety of forms by several types of equipment. The following is a list of typical phenoxy herbicide application techniques in forestry in a decreasing order of occurrence and importance:

Table II. *Use of 2,4,5-T on National Forests*[a], *fiscal years 1971–1978*[b].

Fiscal year	Pounds	Acres
1971	57,402	25,777
1972	40,501	32,100
1973	53,220	26,424
1974	26,022	37,436
1975	95,540	49,357
1976	102,720	53,251
1977	14,597	7,105
1978	6,841	4,167

[a] Land administered by the USDA Forest Service as part of the National Forest System. 1 lb = 0.454 kg, 1 A = 0.405 ha.
[b] Derived from USDA Forest Service Pesticide Use Advisory Memorandum 210, October 26, 1978.

1. Aerial application of liquids by helicopter.
2. Ground application of liquids by high-volume, low-pressure sprayers or low-volume, high-velocity mist blowers.
3. Injection or basal application of liquids into or on individual tree stems.

Applications may be made at almost any time of the year, but dormant season applications just before or at bud break and foliage applications as full foliage expansion occurs or at the end of the growing season are most common. The formulations used and the methods, time, and rate of application all depend on the management objective and the density and species composition of the vegetation (NEWTON and NORRIS 1980).

III. The behavior of phenoxy herbicides in the forest

Phenoxy herbicides applied in the forest are distributed initially in the air, vegetation, forest floor, and surface waters. The exact proportion of spray material entering any of these four environments will vary with chemical factors (herbicide formulation and carrier), application factors (rates and volumes of application and the type of application equipment and its operating characteristics), climatological factors (wind speed and direction, relative humidity, and temperature), and several site factors (slope, aspect, elevation, vegetation type and density, and proximity to surface water). This concept of the initial distribution of herbicide in the forest is intuitive because there is only a limited data base. Some studies report the concentration of herbicide in various forest compartments immediately after or at intervals after application, but mass-balance studies are lacking. The following sections consider the movement, persistence, and fate of the phenoxy herbicides in each of the main compartments of the forest.

a) Behavior in air

Phenoxy herbicides may be in the air as vapors or in fine droplets which do not settle in the target area. They may settle or be washed down by rain into adjacent areas, adsorb on a number of surfaces, or be decomposed by photochemical reactions. Little work in this area has been done in the forest.

The appearance of "box elder blight" in the early 1950s was one of the first symptoms in the forest of a problem which also plagued agriculture, *i.e.*, the offsite movement of growth-regulator chemicals (like the phenoxy herbicides). PHIPPS (1963 and 1964) conducted studies which showed 2,4-D was the probable causative agent, although his test could not rule out the possibility that other phenoxy herbicides (or growth-regulating chemicals) might also be involved. He noted abnormal growth characteristics on elms (*Ulmus* sp.) and ash (*Fraxinus* sp.) could be

caused by 2,4-D. The 2,4-D source causing box elder blight is not known for certain, but agricultural use was implicated. The *National Research Council of Canada* (1978) monograph on phenoxy herbicides gives an extensive review of the data on herbicides in air from agricultural use. The fate of pesticides in the atmosphere has been reviewed by several authors with considerable attention to concepts and some data, but the emphasis has been on insecticides, and the phenoxy herbicides have received little attention (WHEATLEY 1973, SEIBER *et al.* 1975, MOILANEN *et al.* 1975, SPENCER and CLIATH 1975, CREWS *et al.* 1975).

In the forest, NORRIS (1967), citing unpublished data of Norris, Newton, and Zavitkovski, reported on the application of low-volatile esters of 2,4,5-T in diesel oil by fixed-wing aircraft in a western Oregon forest. They found this application resulted in the deposit of 25 to 40% of the chemical at ground level in small openings in the forest canopy. The implication is that 60 to 75% of the herbicide was lost from the application zone. MAKSYMIUK (1963), however, reported more than 40% of aerially applied insecticide did not reach ground level when spray cards were within one tree-height of the nearest vegetation.

Fixed-wing applications of herbicide are not typical in forests. More commonly, applications are made by helicopters which fly relatively slowly and low with a resulting increase in herbicide deposit in the intended area. NORRIS *et al.* (1976 c and 1981), using filter paper interception discs, recivered 85 and 70% of helicopter-applied 4-amino-3,5,6-trichloropicolinic acid (picloram) and 2,4-D from spray interception discs located above the brush. Spray interception discs yielded recoveries of 71% for 2,4-D and 90% for picloram applied by helicopter in four power-line rights-of-way in forested areas of Oregon and Washington (NORRIS *et al.* 1976 a).

Only limited data are available on drift or deposition after application by mist blower. In a test done under nearly ideal conditions, NORRIS *et al.* (1976 d) reported 92% of the picloram and 97% of the 2,4-D applied by a large truck-mounted mist blower was deposited in a 32.9-by-67-m test grid. BRADY and WALSTAD (1973) found 2,4,5-T at distances up to 150 m downwind from a sled-mounted mist blower based on herbicide deposit on spray cards or the response of herbicide-sensitive plants. Soybeans more than about 70 m from the line of application were not killed. HOLT *et al.* (1978) also studied mist-blower applications. They reported effects on plants at 440 m, but the spray mixture used contained both picloram and 2,4,5-T. At 240 m, epinasty was noted on sensitive plants suspended at 3.3-m intervals up to 12 m above the ground. With ground equipment in Texas, SCIFRES *et al.* (1977) reported 92% of the intended spray deposit (2,4,5-T and picloram) was recovered on mylar cards at ground level.

Both fine spray particles (drift) and vapors of phenoxy herbicide esters may move from target areas during and shortly after application. There is little data from the forest environment. In an agricultural setting

Grover *et al.* (1972) applied butyl ester and dimethylamine salt formulations of 2,4-D by ground equipment that resulted in only 2.8% of the total spray having a particle size less than 200 µm and only 3 to 4% of the spray drifted. However, 25 to 30% of the butyl ester was collected as vapor 75 m downwind 30 min after application. The butyl ester is substantially more volatile than the esters [2-ethylhexyl, butoxyethanol, isooctyl, propylene glycol butyl ether (PGBE)] used in forestry, but these findings indicate more studies of phenoxy herbicide ester volatility in the field are needed.

Spray drift was monitored in connection with an operational aerial application of 2,4-D low-volatile ester on the Chippewa National Forest in Minnesota. The amount of 2,4-D deposited on exposed trays 15 m from the treatment unit boundary was 50 to 90% less than the deposit inside the unit. This suggests only limited offsite movement of herbicide occurred. Residues ranged from 0.0005 to 0.1 kg/ha at 15 m (*Minnesota Department of Natural Resources* 1977).

These reports do not indicate actual residue levels in air, but they indicate that residues are present (at least for short periods of time) at concentrations sufficient to cause effects on sensitive plant species. Lavy (1979) measured 2,4,5-T in air as part of a study to evaluate applicator exposure. Only those workers involved with application by backpack sprayer or tractor-mounted mist blower received exposure to measurable amounts of 2,4,5-T via the air. The concentration in air ranged from not detectable (<0.004 µg/L) to 0.169 µg/L; the mean value was 0.02 µg/L. Cheney *et al.* [not dated] used motorized samplers and a resin to monitor 2,4,5-T residues in the air at intervals after aerial application of 3.36 kg/ha 2,4,5-T as the PGBE ester. They found detectable residues in air within the sprayed area for three wk after application. In general the levels were higher in samples collected between 1200 and 1400 hr than between 0700 and 0900 hr. The residues were highest the first 24 hr after application and then were substantially lower. No residues were detected 32 days after application (Table III). Presumably, the residue levels in air decline rapidly after application ceases because (1) fine particles settle out, (2) vapor production slows or ceases as ester hydrolysis occurs, and (3) the air mass is exchanged with fresh air.

The paucity of data on phenoxy herbicide residues in air in connection with forest spray operations indicates this is a significant gap in knowledge. Improving theory and methodology and a more extensive literature on this topic for agricultural situations suggest future studies in the forest will be easier to conduct and interpret.

b) Behavior in vegetation

The behavior of phenoxy herbicides in vegetation is discussed in three sections. The first and second sections consider the initial distribution and concentration of chemical in the canopy as a means of establishing

Table III. *Levels of 2,4,5-T in air after application of 3.36 kg/ha 2,4,5-T as PGBE ester.*[a]

Time after application (days)	Time of day	2,4,5-T ester ($\mu g/m^3$)
0	1000–1200	faulty measure
	1200–1400	0.679
1	0700–0900	0.117
	1200–1330	0.895
3	0650–0850	0.033
	1200–1400	0.175
8	0700–0900	0.012
	1200–1400	0.029
15	0700–0900	0.017
	1200–1400	0.017
21	0700–0900	<0.008
	1200–1400	0.054
32	0700–0900	<0.008
	1200–1400	<0.008

[a] CHENEY *et al.* [not dated].

maximum initial residue levels which are likely to occur during or shortly after application. The third section covers the persistence characteristics of the phenoxy herbicides to establish the residue levels as they change over time. Nearly all the data in both sections are from actual or simulated aerial application of herbicide. Little work has been done to characterize the distribution patterns which result from broadcast or selective ground application systems or procedures.

1. Initial distribution in the canopy.—The amount of the herbicide intercepted by vegetation varies with the rate and nature of the application and the type and density of the vegetation. ALTOM and STRITZKE (1972) found 33% of aerially applied 2,4-D, dichlorprop, and 2,4,5-T penetrated the overstory, and 13% reached the forest floor in a blackjack (*Quercus marilandica*) and a post oak (*Quercus stellata*) forest. The implication is that 20% of the material was intercepted by understory vegetation. In a similar forest type BOUSE and LEHMAN (1967), using nontoxic spray materials, found overstory penetration of 19 to 22% while penetration through the understory was 4 to 6%, indicating about 15% interception. TSCHIRLEY (1968) found about 21% of aerially applied phenoxy herbicides penetrated the upper canopy and 6% penetrated to ground level in tropical and subtropical forests. The percent spray penetration was inversely related to canopy density.

2. Concentration in vegetation immediately after application.—NORRIS *et al.* (1977) looked at the initial distribution of 2,4,5-T low-volatile ester in oil applied by helicopter to a mixed hardwood brush

community in northwest Oregon. They found marked contrast in the concentration of 2,4,5-T among various species shortly after application which indicates the nature of the intercepting surface is also important (Table IV). For instance, 11 mg/kg of 2,4,5-T was on vine maple (*Acer circinatum*) compared to 120 mg/kg on grass. The vine maple buds were swollen but foliage had not yet expanded, thus they had a relatively small surface-to-mass ratio at the time of application. Grass, on the other hand, had a large surface area relative to its mass, and the concentration of herbicide was high.

When blackberry foliage (*Rubus* sp.), grass, and vine maple were re-treated one yr later, the initial herbicide concentrations were higher (Table IV). This reflects a general decrease in the density of overstory vegetation from the yr before. The initial concentration on Douglas-fir (*Pseudotsuga menziesii*) was about the same in both yr, probably because the individual trees sampled were growing in the open and the overstory density did not change from one year to the next.

In a study in West Virginia, Norris *et al.* (1978) found initial residue levels of 2,4,5-T on four kinds of vegetation ranging from 300 mg/kg on blackberry foliage to 80 mg/kg on greenbriar (Table IV) after helicopter application of 2.24 kg/ha ae as butoxyethanol ester. Wellenstein (1975 a and b) measured residues of 2,4,5-T on foliage of several species after helicopter application of ester formulation in Europe. Extrapolating from a graph, initial concentrations ranged from 40 mg/kg on hazel to 500 mg/kg on oak and spruce (Table IV). These values for oak and spruce presumably reflect their dominant position in the canopy, while the lower values for birch (60 mg/kg) and hazel more nearly reflect the levels reported by others for understory vegetation. The initial residue levels are difficult to interpret because the rate of application was not specified.

Plumb *et al.* (1977) reported initial herbicide concentrations of 95 and 92 mg/kg of 2,4-D and 2,4,5-T, respectively, in chamise (*Adenostoma fasiculatum*) immediately after a simulated aerial application of 4.48 kg/ha ae of 2,4-D and 2,4,5-T in southern California (Table IV). Radosevich and Winterlin (1977) reported about 19% of the spray material was on the overstory chamise plants immediately after ground-sprayer application of 4.5 kg/ha of 2,4-D or 2,4,5-T (as butoxypropyl ester). About 28% was on understory grass and forbs and 52% was on the litter (forest floor material). Radosevich and Winterlin did not report actual residue values, but by calculation from their Table 2, initial residue levels were about 200 mg/kg on chamise and 300 mg/kg on grass and forbs immediately after application.

Eronen *et al.* (1979) reported on the behavior of MCPA in a forest ecosystem. Residues of 344 mg/kg were found in birch twigs and leaves and 74 mg/kg in moss collected immediately after aerial application of 2.5 kg/ha of MCPA isooctyl ester (Table IV).

Grass is an important component of both forest and range ecosystems. Grass communities have potential for high herbicide concentrations be-

cause they are a relatively low-growing type of vegetation with a large surface-to-mass ratio. BOVEY and BAUR (1972) applied 2,4,5-T to a grass community at five locations in Texas and found initial herbicide residues ranged from 32 to 144 mg/kg for rates of application of 0.56 and [illegible] kg/ha ae (Table IV). MORTON *et al.* (1967) reported herbicide residue levels in grass ranged from 20 to 210 mg/kg (Table IV) immediately after simulated aerial applications of several formulations of 2,4,5-T and 2,4-D. Immediately after application of 2,4,5-T and picloram (1.12 kg/ha each) with ground equipment to range sites in Texas, SCIFRES *et al.* (1977) found 2,4,5-T residues in coastal bermuda grass ranged from 58 to 125 mg/kg (Table IV).

OLBERG (1973) and OLBERG *et al.* (1974), concerned about 2,4,5-T residues in wild raspberry fruits (species not identified), reported that initial herbicide concentrations ranged from 0.7 to 3.3 mg/kg 1 hr after treatment with 6 kg 2,4,5-T in 250 L water/ha in tests conducted in 1972. Apparently similar applications in 1973 produced initial 2,4,5-T residues ranging from 7.9 to 22.2 mg/kg. Applications in both cases were by "backpack power sprayer." In contrast, MAIER-BODE (1972) found only 1 mg/kg of 2,4,5-T on unidentified wild berries in Sweden on the day of treatment by aircraft. BROWN and MACKENZIE (1971) found residues of 2,4,5-T up to 100 mg/kg on unwashed blackberry fruits 3 days after spraying with 3 kg/ha of 2,4,5-T. Ripe fruits remained in good condition for about 3 days, but immature fruits failed to ripen.

These reports indicate initial phenoxy herbicide residues in forest vegetation can range up to about 300 mg/kg for rates of application up to 2.24 kg/ha ae. Proportionally higher residue levels may be expected for higher rates of application.

3. **Persistence characteristics.**—Phenoxy herbicide residues decline with time in forest vegetation through the action of several processes including volatilization, photochemical or biological degradation on surfaces, weathering (rain washing, cuticle erosion), absorption and translocation, growth dilution, metabolism, excretion, and others. Most field studies only determine residue levels and do not determine the importance of specific residue-reduction processes. The following paragraphs provide specific data on phenoxy herbicide residue levels in forest vegetation as a function of time after application. There are more data for 2,4,5-T than for 2,4-D. There are few reports on the persistence of other phenoxy herbicides in forest vegetation.

PLUMB *et al.* (1977) simulated the aerial application of 2,4-D and 2,4,5-T (PGBE ester, 4.48 kg ae each in 186 L of water/ha) to a 3-yr-old stand of chamise in southern California. 2,4-D and 2,4,5-T had half-lives of about 37 and 17 days in this vegetation (Table IV). The rate and extent of decline of these herbicide residues was not as great as is noted in some other studies, probably because the site was very dry. Plant moisture levels were about half of normal at the time of application and declined to less than 30% 9 wk after the application. This largely eliminated metabolism of the residues. About 3 mg/kg of 2,4-D and

2,4,5-T were present in the dead vegetation 1 yr after application. Sprouts from the treated plants did not show formative effects but did contain 0.27 mg/kg 2,4-D and 0.31 mg/kg 2,4,5-T 1 yr after application. These plant parts were not present at the time of application, indicating these residues resulted from the translocation of chemical from treated portions of the plant.

RADOSEVICH and WINTERLIN (1977) also measured 2,4-D and 2,4,5-T residues in chapparal vegetation. They expressed their data relative to the total amount of herbicide recovered in vegetation, litter, and soil immediately after application of 4.5 kg/ha of each herbicide. They found residues declined rapidly (more than 90%) in the first 30 days after application, but then the rate of loss declined. Calculations based on their Table 2 suggested residues on vegetation were 0.1 to 0.2 mg/kg 1 yr after application.

NORRIS *et al.* (1977) determined residues of 2,4,5-T in 4 species of forest vegetation after two successive annual applications (4.48 kg/ha of 2,4,5-T ae as isooctyl ester applied in diesel oil by helicopter in March). A sharp decrease in herbicide concentration occurred the first mon after application, but the rate of residue disappearance slowed after 3 mon (Table IV). The mean half-life of 2,4,5-T for all species was about 2 wk after both the first and the second application. One yr after application, residues ranged from 0.48 mg/kg in vine maple to 0.07 mg/kg in blackberry foliage. On 3 plots which were not sprayed the second time, 2,4,5-T herbicide residues were below detectable limits in all species except vine maple 2 yr after the first application. The rate of decline of 2,4,5-T residues in vegetation after the second application was similar to the first except that 1 yr after the second application, no residues were detected in any of the sprayed vegetation. Thus, in this case at least, 2 successive annual applications of 2,4,5-T had no appreciable effect on the persistence of the herbicide in 4 different kinds of vegetation.

In West Virginia, residues of 2,4,5-T on 4 species declined rapidly after application (2.24 kg/ha 2,4,5-T ae as butoxyethanol ester applied in August by helicopter). The mean concentration was 150 mg/kg immediately after application and 25 mg/kg at 1 mon, 5 mg/kg at 6 mon, 0.07 mg/kg at 1 yr, and <0.01 mg/kg at 2 yr (Table IV). The initial rate of decline was slower in West Virginia than in a comparable study in Oregon (NORRIS *et al.* 1978).

WELLENSTEIN (1975 a and b) reported relatively rapid decline (90+% in 10 wk) in concentration of herbicide in foliage of 4 species of woody plants in a European forest treated with 2,4,5-T ester by helicopter. The rate of loss was slower after 10 wk, and residues ranged from 0.25 to 0.4 mg/kg 1 yr after application (Table IV). ELIASSON (1973) applied butoxyethyl ester of 2,4-D to young aspen trees (*Populus* sp.) in a glasshouse experiment and found a marked decrease in herbicide residue level with time, despite the fact an extremely high concentration of herbicide was present initially, and more than half the sprayed leaf tissue was dead after 9 days (Table IV).

Table IV. *Phenoxy herbicide residues in vegetation.*

Herbicide	Location	Plant species	Application	Residue level in mg/kg (days after application)	References
2,4-D	So. Calif.	Chamise	3.4 kg/ha ae PCBE ester in water, simulated aerial application, May	95(0) 70(14) 69(29) 20(69) 16(146) 3.8(379)	PLUMB *et al.* (1977)
2,4-D	So. Calif.	Chamise	4.5 kg/ha ae butoxy propyl ester in water, ground sprayer, April and May	221(0) 60(30) 35(60) 22(90) 12(180) 0.1(360)	RADOSEVICH & WINTERLIN (1977)
2,4-D	So. Calif.	Grass and forbs	4.5 kg/ha ae butoxy propyl ester in water, ground sprayer, April and May	269(0) 21(30)	RADOSEVICH & WINTERLIN (1977)
2,4-D	Texas	Grass	1.12 kg/ha ae 2,4-D amine in water, simulated aerial application, June	80(0) 70(7) 45(14) 30(28) 6(56) 1(112)	MORTON *et al.* (1967)
2,4-D	Sweden	Poplar	Glass house application, 2,4-D butoxyethyl ester in diesel oil	2300(1) 2500(3) 1800(9) 1300(37) 870(365)	ELIASSON (1973)
2,4-D	Sweden	Cowberry and bilberry	0.25 kg/ha ester, ground application	0.3(34) and 0.1(29)	RAATIKAINEN *et al.* (1979)
			0.75 kg/ha	1.0(35) and 1.3(29)	
			2.25 kg/ha	3.7(35) and 4.8(29)	
2,4,5-T	Texas	Grass	1.12 kg/ha ae PCBE ester in water, simulated aerial application	73(0) 2.1(42) 0.02(182)	BOVEY & BAUR (1972)
2,4,5-T	Sweden	Cowberry and bilberry	0.25 kg/ha ester, ground application	0.5(34) and 0.4(29)	RAATIKAINEN *et al.* (1979)
			0.75 kg/ha	1.2(35) and 1.3(29)	
			2.25 kg/ha	3.9(35) and 2.5(29)	
2,4,5-T	Germany	Raspberry (fruits)	6 kg/ha formulation not known, in water, foliage application from ground, June	16(0) 11.2(5) 3.4(15) 1.5(30) (by interpolation, Table 2)	OLBERG *et al.* (1974)

2,4,5-T	Texas	Live oak (stem tips)	2.24 kg/ha ae 2,4,5-T isooctyl ester in water, simulated aerial application, June	9.6(30) 0.7(180)	Baur *et al.* (1969)
2,4,5-T	Texas	Grass	2.24 kg/ha ae 2,4,5-T isooctyl ester in water, simulated aerial application, June	7.0(30) 0.2(180)	Baur *et al.* (1969)
2,4,5-T	Texas	Coastal bermuda grass	1.12 kg/ha ae (formulation not specified) in 127 L/ha of diesel oil:water (1:4) with ground sprayer	246(0) 7.9(7) 4.9(28) 0.42(56) 0.05(112) 0.005(365) (average values from Table 1)	Scifres *et al.* (1977)
2,4,5-T	Europe	Raspberry fruits	6 kg/ha Tormona salt in water, backpack sprayer	8.3(0) 8.1(4) 7.8(7)	Eichler & Leber (1975)
2,4,5-T	Europe	Raspberry foliage	6 kg/ha Tormona salt in water, backpack sprayer	368(0) 19(7) 28(13) 15(21) 5(36) 4(51) 2(83)	Eichler & Leber (1975)
2,4,5-T	Europe	Grass	6 kg/ha Tormona salt in water, backpack sprayer	61(0) 8(7) 4(13) 3(21) 0.8(36)	Eichler & Leber (1975)
2,4,5-T	Europe	Blackberry fruits	2.5 L/ha Tormona 80 in water, helicopter	7.7(0) 8.3(4) 7.5(7)	Eichler & Leber (1975)
2,4,5-T	Texas	Grass	0.56 kg/ha ae 2,4,5-T butoxyethanol ester in water, simulated aerial application, June	48(0) 35(7) 10(14) 9(28) 7(56)	Morton *et al.* (1967)
2,4,5-T	Texas	Grass	2.24 kg/ha ae 2,4,5-T butoxyethanol ester in water, simulated aerial application, June	205(0) 150(7) 50(14) 60(28) 25(56)	Morton *et al.* (1967)
2,4,5-T	Oregon	Douglas-fir	2.24 kg/ha isooctyl ester in oil, helicopter application in early spring		Norris *et al.* (1977)
			First annual application	52(0) 11.1(30) 0.35(90) 0.47(180) 0.22(360) 0.0(720)	
			Second annual application	52(0) 14.2(30) 0.10(90) 0.04(180) 0.0(360)	

Table IV. (*Continued*)

Herbicide	Location	Plant species	Application	Residue level in mg/kg (days after application)	References
2,4,5-T	Oregon	Vine maple	First annual application	10.6(0) 0.48(30) 0.28(90) 0.16(180) 0.48(360) 0.02(720)	NORRIS *et al.* (1977)
			Second annual application	23.2(0) 10(30) 0.10(90) 0.10(180) 0.02(360)	
2,4,5-T	Oregon	Grass	First annual application	114(0) 3.4(30) 0.58(09) 0.14(180) 0.12(360) 0.0(720)	NORRIS *et al.* (1977)
			Second annual application	140(0) 9.3(30) 0.21(90) 0.12(180) 0.0(360)	
2,4,5-T	Oregon	Blackberry (vines & foliage)	First annual application	45(0) 0.59(30) 0.05(90) 0.02(180) 0.03(360) 0.0(720)	NORRIS *et al.* (1977)
			Second annual application	165(0) 2.9(30) 0.01(90) 0.0(180) 0.0(360)	
2,4,5-T	So. Calif.	Chamise	3.4 kg/ha ae, PGBE ester in water, simulated aerial application, May	92(0) 44(14) 32(29) 14(69) 2.9(140)	PLUMB *et al.* (1977)
2,4,5-T	So. Calif.	Chamise	4.5 kg/ha ae PGBE ester in water, ground sprayer, April and May	184(0) 13(30) 8(60) 10(90) 8(180) 0.2(360)	RADOSEVICH & WINTERLIN (1977)
2,4,5-T	So. Calif.	Grass and forbs	4.5 kg/ha ae PGBE ester in water, ground sprayer, April and May	319(0) 31(30)	RADOSEVICH & WINTERLIN (1977)
2,4,5-T	W. Virginia	Blackberry (foliage)	2.24 kg/ha ae butoxyethanol ester in water, helicopter application in August	289(0) 156(7) 48(30) 8.8(90) 3.4(182) 0.01(365) 0.003(730)	NORRIS *et al.* (1978)

2,4,5-T	W. Virginia	Greenbriar	2.24 kg/ha ae butoxyethanol ester in water, helicopter application in August	84(0) 28(7) 9.7(30) 4.3(90) 1.0(182) 0.03(365) 0.002(730)	Norris *et al.* (1978)
2,4,5-T	W. Virginia	Sassafras	2.24 kg/ha ae butoxyethanol ester in water, helicopter application in August	141(0) 33(7) 19(30) 8.1(90) 2.2(182) 0.016(365) 0.003(730)	Norris *et al.* (1978)
2,4,5-T	W. Virginia	Grass	2.24 kg/ha ae butoxyethanol ester in water, helicopter application in August	91(0) 28(7) 17(30) 11(90) 12(182) 0.22(365) 0.03(730)	Norris *et al.* (1978)
2,4,5-T	Europe	Spruce	Unspecified rate of ester by helicopter	500(0) 20(70) 1.5(210) 0.4(350) 0.1(665)	Wellenstein (1975 a and b)
2,4,5-T	Europe	Birch	Unspecified rate of ester by helicopter	60(0) 8(70) 1(210) 0.25(350) 0(665)	Wellenstein (1975 a and b)
2,4,5-T	Europe	Oak	Unspecified rate of ester by helicopter	500(0) 1.5(70) 0.4(210) 0.25(350) 0.1(665)	Wellenstein (1975 a and b)
2,4,5-T	Europe	Hazel	Unspecified rate of ester by helicopter	40(0) 5(70) 0.9(210) 0.32(350) 0.05(665)	Wellenstein (1975 a and b)
2,4,5-T	California	*Ceanothus* sp.	3.36 kg/ha as PGBE ester, helicopter application in October	147(0) 154(1) 95(3) 55(8) 10.5(15) 4.8(21) 8.5(32) 0.9(228) (combined ester and acid forms)	Cheney *et al.* [not dated]
MCPA	Sweden	Cowberry and bilberry	Ground application		
			0.25 kg/ha ester	0.2(34) 0.2(29)	Raatikainen *et al.* (1979)
			0.75 kg/ha ester	0.8(35) 0.7(29)	
			2.25 kg/ha ester	3.0(35) 3.9(29)	
MCPA	Finland	Birch leaves and twigs	Aerial application 2.5 kg/ha isooctyl ester	344(0) 119(15) 132(40) 25(261) 32(283) 44(297)	Eronen *et al.* (1979)
MCPA	Finland	Moss	Aerial application 2.5 kg/ha isooctyl ester	74(0) 17(15) 3(40) 0(283)	Eronen *et al.* (1979)

2,4,5-T was applied at 2.24 kg/ha as the 2-ethylhexyl ester alone or in combination with picloram to live oak (*Quercus virginia*) and nati[illegible] grasses by BAUR *et al.* (1969). They did not collect samples immed[illegible], after application but did show that 90 to 99% of the 2,4,5-T prese[illegible] in samples collected 1 mon after the treatment were no longer present in samples collected 6 mon after treatment. Persistence of 2,4,5-T was longer in plants treated with 2,4,5-T-picloram mixture than in plants treated with 2,4,5-T alone (Table IV). BOVEY and BAUR (1972) also looked for 2,4,5-T in range grass in Texas after application of herbicide at 0.56 and 1.12 kg/ha ae as PGBE ester in 186 L of water/ha in a simulated aerial application. A 98% reduction in 2,4,5-T levels was found 6 wk after application with nondetectable or nearly nondetectable levels reported 26 wk after application. Rainfall influenced the rate of loss of 2,4,5-T at different sites in this study. MORTON *et al.* (1967) found a sharp decrease in the levels of 2,4-D and 2,4,5-T in range grass treated with ester, acid, and amine formulations of these herbicides in simulated aerial applications at rates of 0.56, 1.12, or 2.24 kg/ha ae (Table IV). They reported no effect of formulation on persistence but that amount and frequency of rainfall were important. On coastal Bermuda grass treated with 1.12 kg/ha each of 2,4,5-T and picloram, 92% of the 2,4,5-T present immediately after spraying was gone in 7 days (Table IV) (SCIFRES *et al.* 1977). RADOSEVICH and WINTERLIN (1977), on the other hand, indicated rainfall 30 days after application was not a major factor in residue reduction on leaves. More than 90% of the herbicide had already disappeared from foliage in their study by 30 days. If the rain had reduced remaining residues by 50%, this loss may not have been easily detected.

Phenoxy herbicide esters are believed to hydrolyze rapidly on plant surfaces. SUNDSTROM *et al.* (1979) looked for both acid and ester forms of 2,4,5-T in foliage from brush vegetation treated by tractor sprayer (2 to 5 kg/ha ae 2-butoxyethyl ester) or airplane (1.5 to 3 kg/[illegible] isobutyl, butyl ester). Results were highly variable for samples from the tractor-sprayed areas reflecting the uneven initial distribution of herbicide. They found 640 mg/kg of 2-butoxyethel ester and 390 mg/kg of acid of 2,4,5-T 8 days after application. After 45 days, the levels were 180 mg/kg for both. Residues were generally lower in the aerially treated area. They averaged 38 mg/kg of butyl ester, 16 mg/kg of isobutyl ester, and 61 mg/kg of acid of 2,4,5-T 18 days after application. The rate of ester hydrolysis apparent in this study was not as rapid as has been reported by GLASTONBURY *et al.* (1958), CRAFTS (1960), and SZABO (1963). SUNDSTROM *et al.* (1979) cited detection of the 2-butoxyethyl ester of three different dichlorophenoxy isomers as evidence of the photochemical degradation of the 2,4,5-T.

CHENEY *et al.* [not dated] reported 2,4,5-T ester residue levels decreased rapidly on foliage of *Ceanothus* sp. after aerial application of 3.36 kg/ha as PGBE ester (Tables IV and V). They noted a 98% reduction in the level of ester and a 61% reduction in the level of acid the

Table V. *Levels of 2,4,5-T on foliage of Ceanothus sp. after aerial application of 3.36 kg/ha 2,4,5-T as PGBE ester.*[a]

Time after application (days)	Concentration of 2,4,5-T (mg/kg)	
	Ester	Acid
0[b]	132	15
1	142	12
3	90	5.1
8	51	4.1
15	6.3	4.2
21	3.0	1.8
32	2.7	5.8
223	<0.04	0.9

[a] CHENEY *et al.* [not dated].
[b] Immediately after spray.

first 30 days after application. The rapid loss of ester did not appear to be matched by a corresponding rise in the level of acid. These results suggest that while hydrolysis is certainly occurring, other processes (such as volatilization) may also be responsible for loss of the ester from the plant. Their finding of residues in air for several days after the application is consistent with this hypothesis.

Studies of 2,4,5-T residues in raspberry fruits in European forests have provided peculiar data. Based on reports by OLBERG (1973) and OLBERG *et al.* (1974), it appears that 2,4,5-T (6 kg in 250 L of water/ha in June and July) caused relatively fast leaf-wilt, but green berries continued to ripen and became "conspicuously large and beautiful." The results of residue analysis present a confusing picture. Initial residue levels were markedly different in the 2 yr of the study. First-year results with one formulation show a four-fold decrease in residue level in 41 days but virtually no change in residue level over the same period with a second formulation. The second year, initial residue levels were higher than the first year by a substantial margin. These levels declined relatively fast, however, with a mean half-life of 8.6 days for the first 15 to 17 days after treatment. There was a marked reduction in the rate of decrease after that time (Table IV). Residue levels varied from 0.4 to 2.2 mg/kg by the end of the measurement period, which ranged from 29 to 41 days on different plots. These levels are substantially greater than the 0.05 mg/kg residue level permitted in Germany. The results are confounded to some degree by apparent 2,4,5-T residues in untreated fruits. One control set had no detectable 2,4,5-T, but the other three contained residues ranging from 0.14 to 016 mg/kg (3- to 12-times greater than the permissible limit).

The successful development of the fruit after application makes one wonder about the overall effectiveness of the treatment. Some modifica-

tion of formulation, carrier, or technique of application might accomplish more complete early season control such that treated fruits do not ripen. As a result of these studies, the season of application of 2,4,5-T in German forests was restricted to the period before fruit-set or after fruit harvest.

SILTANEN and ROSENBERG (1978) and MUKULA *et al.* (1975 and 1978) analyzed for residues in lingonberry, wild mushrooms, and aspen foliage from northeastern Finland after aerial application of 2,4-D and 2,4,5-T (2.5 kg/ha). The samples were collected at various periods from 2 to 52 wk after application. In lingonberries, residues of 2,4-D varied from <0.1 to 5.6 mg/kg and 2,4,5-T from 0.06 to 15 mg/kg at intervals up to 13 wk after application. Residues were not detected 1 yr later (2,4-D <0.05 mg/kg, 2,4,5-T <0.02 mg/kg) in fruits from plants which had survived the previous year's spraying. Wild mushrooms contained <0.05 to 1.2 mg/kg of 2,4-D and <0.02 to 1.8 mg/kg of 2,4,5-T at intervals up to 8 wk after application. In leaves and twigs of aspen, residues varied from 0.1 to 30 mg/kg of 2,4-D and 0.2 to 30 mg/kg of 2,4,5-T at intervals up to 3 mon after application. Similar values are reported for birch. In lingonberry plants killed by the spray, herbicide residues were still present (3 to 14.5 mg/kg of 2,4-D and 5.5 to 5.7 mg/kg of 2,4,5-T) in dead leaves and twigs 1 yr after application. These findings agree with those of PLUMB *et al.* (1977).

RAATIKAINEN *et al.* (1979) reported on residue levels of 2,4-D, 2,4,5-T, and MCPA in fruits of cowberry (*Vaccinium vitis-idaea*) at 35 days and bilberry (*V. myrtillus*) at 29 days after ground application of ester formulation of the herbicides at rates of 0.25, 0.75, and 2.25 kg/ha active ingredient. The residue levels were quite similar among compounds and between species, and there was an almost straightline correlation between the rate of application and the residue level (Table IV). The general level of the residues was of the same order of magnitude reported by MUKULA *et al.* (1978) for a similar study involving aerial application.

EICHLER and LEBER (1975) looked for residues of 2,4,5-T in honey, wild raspberry and blackberry fruits, wild raspberry leaves, and grass after ground application (backpack sprayer) of 6 kg/ha of 2,4,5-T in water in late July. Residues on vegetation declined more than 90% in 2 wk (Table IV). After 1 wk the fruit had lost its turgidity and was no longer considered desirable for picking. Jelly prepared from these fruits contained 1.8 mg/kg of 2,4,5-T: a 75 to 80% decrease in concentration in comparison to the initial concentration in the fresh fruit. WELLENSTEIN (1975 b) reported 4.15 mg/kg of 2,4,5-T in blackberry fruits 1 wk after helicopter application of herbicide (rate of application not specified). In marked contrast, MUKULA *et al.* (1978) found no difference in the residue level of 2,4-D or 2,4,5-T between raw lingonberries and jam prepared from the same fruits (about 2.1 mg/kg of each herbicide). Analysis of honey collected from a hive in the treated area and from another hive 50 m from the area 10 days after aerial application of 2,4,5-T ester

showed no detectable (<0.005 mg/kg) 2,4,5-T (EICHLER and LEBER 1975). These findings agree with those of OLBERG *et al.* (1974), but WELLENSTEIN *et al.* (1975) reported 2,4,5-T at 0.006 to 0.009 mg/kg in bee honey.

ERNE and HARTMANN (1973) detected phenoxy herbicides at 4.4 mg/kg in lingonberries, 3.8 mg/kg in bilberries, 1.5 mg/kg in raspberries, and 4.5 mg/kg in mushrooms 4 days after application of herbicide at 2 kg/ha. After 1 mon, the residues were 2.9, 2, 1, and 0.4 mg/kg, respectively. Rinsing was not effective in reducing residues in lingonberries but removed up to 25% of the residues from mushrooms. Parboiling reduced residues by an additional 35%. Residues were not found in commercial lingonberry jam or berries.

Residues of MCPA were determined at different levels of the forest in a study area in northern Finland by ERONEN *et al.* (1979). Samples were collected at 0, 15, 40, 261, 283, and 297 days after aerial application of MCPA isooctyl ester (2.5 kg/ha). MCPA decreased 66% in birch twigs and foliage and 77% in mosses in the first 15 days after application (Table IV). Residues in moss decreased to 3 mg/kg in 40 days and were not detected 283 days after application. The rate of decrease was slower in the birch with 132 mg/kg detected at 40 days and 32 mg/kg still present at 283 days. These values are consistent with the values for 2,4-D and 2,4,5-T in birch and aspen reported by SILTANEN and ROSENBERG (1978).

ELIASSON (1973) provided valuable data on 2,4-D residues in foliage of young aspen treated with butoxyethyl ester in diesel oil as a basal treatment or with water-soluble amine in an injection treatment. Basal spraying resulted in much lower concentration of herbicide in foliage than did the injection treatment (Table VI).

BRADY (1973) did not present actual residue levels but did report 2,4,5-T had a half-life of 5.5 wk in loblolly pine (foliage, presumably) (*Pinus taeda*), 6.7 wk in sweet gum (*Liquidambar styraciflua*), 5.8 wk in post oak, and 12.4 wk in red maple (*Acer rubrum*). Slightly less than 2 wk were required for the loss of half the herbicide in the soil. He also

Table VI. *2,4-D residues in foliage from different parts of the crown of basal-sprayed or stem-injected aspen* (ELIASSON 1973).

Treatment	Crown part	Days after treatment 5 (mg/kg)	28 (mg/kg)	91 (mg/kg)
Basal spray	upper	20	12	20
	middle	10	5	4
	lower	0.2	0.3	0.2
Stem injection	upper	300	250	20
	middle	200	150	80
	lower	14	25	4

noted 97 to 99% of the initial dose remained in the treated foliage. NORRIS and FREED (1966 a and b) also found most of the applied dose remained in treated foliage in a glasshouse experiment. They reported 21 and 16% absorption of 2,4-D and 2,4,5-T as 2-ethylhexyl esters by foliage of bigleaf maple (*Acer macrophyllum*). About 95% of absorbed herbicide was [illegible] in the treated leaves 72 hr after application, and 94% of both herbicides was unchanged structurally in 7 days. Clearly, phenoxy herbicide in and on treated foliage is a potential major source of residues for the forest floor.

c) *Behavior in forest floor and soil*

Phenoxy herbicides reach the forest floor during application or later due to the washing action of rain or as residues in and on leaves that fall from treated plants.

1. Transfer to the forest floor.—In a computer simulation of 2,4,5-T behavior, WEBB *et al.* (1975) reported that rainfall can play an important role in moving herbicide residues to the forest floor. Their model used data for 2,4,5-T washed from red alder (*Alnus rubra*) foliage by artificial rain in a greenhouse study. This study showed rain applied 2 days after low-volatile ester of 2,4-D and 2,4,5-T resulted in washoff of 11 and 12% of the applied herbicide, respectively. Rain applied 9 days after treatment, however, removed only 3.3 and 7.5% of the applied 2,4-D and 2,4,5-T, respectively.

NORRIS *et al.* (1977) found a slight increase in the amount of 2,4,5-T in forest floor material between the time of application and 1 mon posttreatment in an Oregon forest in the spring. They attributed the increase to the washing action of rain on overstory vegetation. Rain is cited as an important factor in reducing plant herbicide residue levels (presumably by washing residues from the leaf surfaces to the ground) in studies with range grass (MORTON *et al.* 1967, BOVEY and BAUR 1972), in [illegible] forest (ERONEN *et al.* 1979), and in model ecosystems (VIRTANEN *et al.* 1979). ELIASSON (1973) tested the effect of rain (and other associated environmental parameters) on 2,4-D residue levels on leaves of young aspen plants treated with large quantities of 2,4-D butoxyethyl ester. Residues of 800, 100, and 20 mg/kg of 2,4-D were found on the leaves 11, 74, and 325 days, respectively, after treatment. Plants treated in the same manner but protected from the rain had 2,4-D residues of 2,000, 1,000, and 900 mg/kg at these same times.

NORRIS *et al.* (1978) measured the transfer of 2,4,5-T from the forest canopy to the forest floor in West Virginia. Throughfall precipitation contained detectable amounts of 2,4,5-T for 8 mon after aerial application (2.24 kg/ha ae as butoxyethyl ester) and deposited about 34 mg/m^2 of 2,4,5-T in the forest floor. More than 85% of the transfer occurred the first 20 days after application (99% in 5 mon). Only 6.0 mg/m^2 were transferred in fresh-fall litter the first year and 0.02 mg/m^2 the second

year after application. At this site, a total of 71.3 mg/m² 2,4,5-T entered the forest floor the first year after application (31.3 mg/m² during application, 34 mg/m² in throughfall precipitation, and 6 mg/m² in fresh-fall litter).

2. Movement and persistence in the forest floor and soil.—Phenoxy herbicide residues in the forest floor and soil are subject to volatilization, adsorption, leaching, uptake by plants, and a variety of chemical and biological degradation processes. These same processes operate in the agricultural environment but at different rates. Most studies in the forest, or using substrates from the forest, only examine reductions in herbicide residue levels with time and do not determine quantitatively the importance of particular residue reduction processes.

Norris (1966 and 1970 a) and Norris and Greiner (1967) reported a series of laboratory studies of the persistence of 2,4-D and 2,4,5-T in forest floor material (L, F, and H horizons) from a red alder stand. Using carboxyl ^{14}C-labelled 2,4-D and 2,4,5-T (2.24 kg/ha ae as triethanolamine salts in water), Norris (1966) found more rapid $^{14}CO_2$ release from 2,4-D- than from 2,4,5-T-treated forest floor material (89% for 2,4-D vs. 23% for 2,4,5-T in 315 hr). Norris and Greiner (1967) applied 2,4-D at 3.36 kg/ha to forest floor material from 5 different types of forest vegetation and determined herbicide persistence at intervals for 15 days (Fig. 1). They found statistically significant (but unimportant) differences in 2,4-D persistence among different types of forest floor material. They examined the influence of formulation on persistence and found a purified 2,4-D acid and laboratory-prepared (purified) 2,4-D triethanolamine salt were more readily degraded than commercial isooctyl ester or solubilized acid formulations (Fig. 1). The most striking difference was between the purified and the solubilized 2,4-D acids (the 2,4-D is in exactly the same chemical form in these two formulations). The slower rate of degradation of the solubilized acid was attributed to constituents of formulation which suggests one chemical can affect the rate of degradation of another. In a further examination of this point, they applied 2,4-D isooctyl ester at 3.36 kg/ha ae alone and with diesel oil at 37.4 L/ha or with DDT [1,1,1-trichloro-2,2-bis(*p*-chlorophenyl)ethane] at 1.12 kg/ha. Diesel oil had no effect on the degradation rate of 2,4-D, but a slight stimulation of 2,4-D degradation was noted in the presence of DDT.

Norris (1970 a) examined in more detail the effects of various insecticides and herbicides on the degradation of 2,4-D and 2,4,5-T in red alder forest floor material. The results of this study are in Table VII. The percent recovery of 2,4-D and 2,4,5-T applied at two rates was the same, *i.e.*, the half-life is independent of starting concentration which suggests the rate of degradation follows first-order kinetics. Picloram and 2,4,5-T may cause an initial slowing of the 2,4-D degradation rate, but these differences were gone by the 35th day. The persistence characteristics of 2,4-D in the field probably will not be greatly altered by the simultaneous application of either 2,4,5-T or picloram.

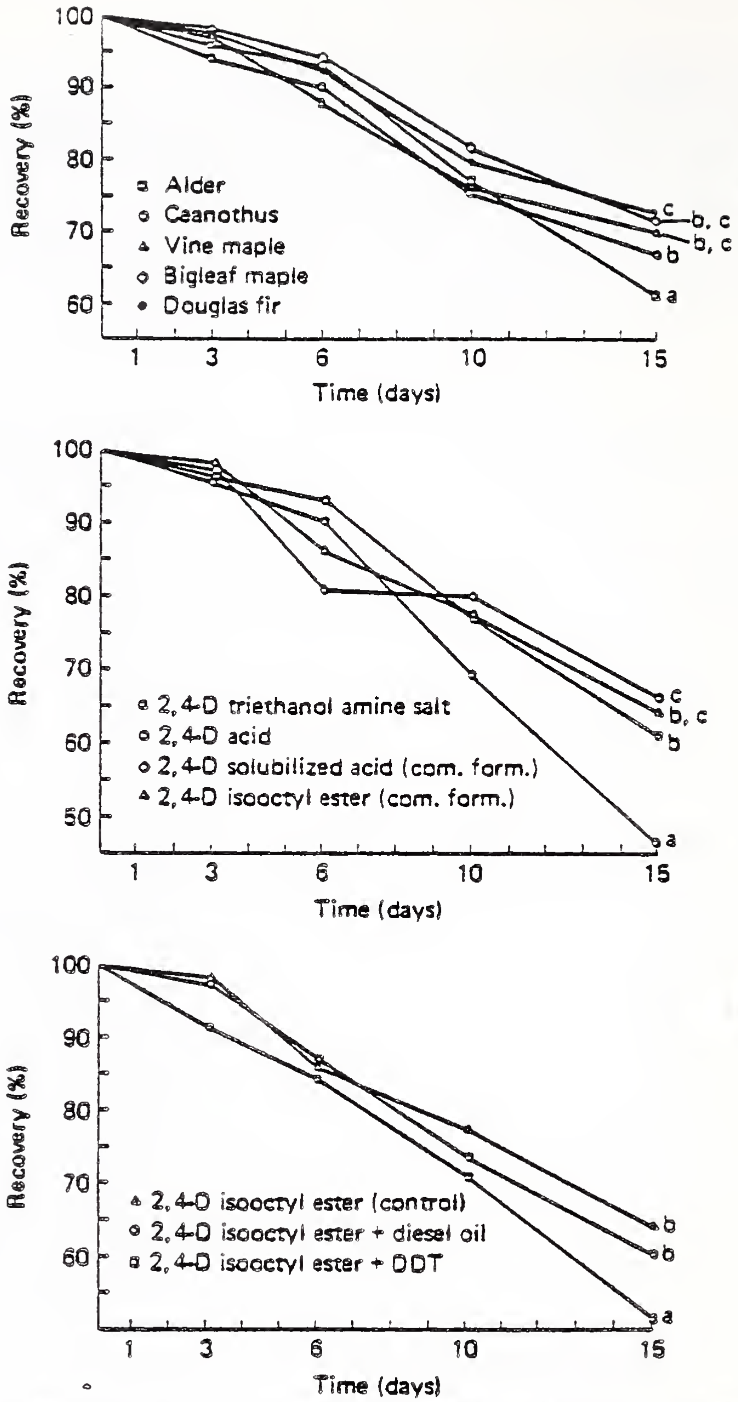

Fig. 1. Recovery of 2,4-D from forest floor material (Norris and Greiner 1967): top = effect of type of forest floor, middle = effect of formulation in red alder forest floor, and bottom = effect of diesel oil and DDT in red alder forest floor.

Table VII. *Recovery of 2,4-D and 2,4,5-T from red alder forest floor material* (NORRIS 1970 a).

Treatment	Recovery of herbicide (%) at		
	10 days	20 days	35 days
2,4-D (2.24 kg/ha)	56	14	6
2,4-D (4.48 kg/ha)	65	16	5
2,4-D (2.24 kg/ha) with 2,4,5-T (2.24 kg/ha)	58	21	8
2,4-D (2.24 kg/ha) with picloram (0.56 kg/ha)	62	14	5
2,4-D (2.24 kg/ha) 1 mon after DDT (1.12 kg/ha)	45	19	9
2,4-D (2.24 kg/ha) 1 mon after phosphamidon (1.68 kg/ha)	55	20	6
2,4-D (2.24 kg/ha) 1 mon after carbaryl (2.24 kg/ha)	51	15	11
	20 days	60 days	120 days
2,4,5-T (2.24 kg/ha)	66	35	13
2,4,5-T (4.48 kg/ha)	78	42	18
2,4,5-T (2.24 kg/ha) with 2,4-D (2.24 kg/ha)	78	24	13

In another test, the forest floor was pretreated with a representative from one of three classes of insecticide: an organochlorine compound, DDT; a carbamate, carbaryl (1-napthyl-N-methylcarbamate); and an organophosphate, phosphamidon (2-chloro-2-diethylcarbamoyl-1-methylvinyl dimethylphosphate). No significant differences were found among 2,4-D recoveries at 35 days except between 2,4-D alone and 2,4-D plus carbaryl. The small reduction in 2,4-D degradation (after application of carbaryl) is probably not important in field use. Thus, it appears the potential for alteration of 2,4-D persistence when applied to areas previously treated with 1 of these 3 insecticides is small. This relationship is also probably true for many other specific members of the organochlorine, carbamate, or organophosphate classes of insecticides and can likely be extended to the other phenoxy herbicides as well.

The rate of degradation of 2,4,5-T was accelerated by the presence of 2,4-D up to 60 days after application, but after 4 mon, recovery was the same as from forest floor material receiving 2,4,5-T alone. These various tests indicate that while it is possible for one chemical to influence degradation rate of another, the magnitude to which this occurs is not likely to result in important changes in the persistence of the phenoxy herbicides as they are used in the field.

In a laboratory study, ALTOM and STRITZKE (1973) studied the degradation rates of 2,4-D, dichlorprop, 2,4,5-T, and silvex in a mixture of soil and litter from two forest sites and one grassland site in Oklahoma. The phenoxy herbicides were applied as diethanol amine salts in water to achieve a concentration of approximately 4.8 mg/kg ae in the soil-litter mixture. The degradation of 2,4-D was rapid in all soils with approximately 0.1 mg/kg remaining 20 days after application. Degradation of

dichlorprop was not as rapid, and average residue levels for the 3 types of soil were 0.46 mg/kg at 20 days and 0.21 mg/kg at 40 days after application. Using average data for the 3 types of soil, 2,4,5-T and silvex showed a similar relative rate of degradation with 2.03 mg/kg remaining at 20 days and 1.05 mg/kg at 40 days. Silvex residues were 1.12 mg/kg and 0.6 mg/kg at these same times (Table VIII).

FOSTER and MCKERCHEN (1973) incubated 2,4-D, MCPA, and 2,4,5-T in several Saskatchewan soils in the laboratory. The reported half-lives varied from 14 to 41 days for 2,4-D and 1 to 2 days longer for MCPA. The half-life of 2,4,5-T was more than twice as long. SMITH (1978) in a similar study added phenoxy herbicides to a prairie soil to achieve 2 ppm concentration and then incubated the soil at 85% field capacity and 20°C in the laboratory. Degradation was rapid in all soils, and the average half-lives for 2,4-D, 2,4-DB, dichlorprop, 2,4,5-T, and silvex were <7, 7, 10, 12, and 12 days, respectively.

Phenoxy herbicides are frequently applied as esters (which have substantially different properties than the acid). The rate of ester hydrolysis to the acid is important in determining the behavior of the herbicide in the soil. SMITH (1972 and 1976) reported rapid hydrolysis of several esters of various phenoxy herbicides. Generally the rate of ester hydrolysis decreased with decreasing soil moisture and with increasing weight of the alcohol moiety. In soils at the wilting point, 90 to 100% hydrolysis of isopropyl, *n*-butyl, and isooctyl esters of 2,4-D occurred in 24 to 48 hr. The same results were found with esters of 2,4,5-T, dichlorprop, and 2,4-DB. BURCAR *et al.* (1966) reported complete disappearance of the isooctyl ester of 2,4-D in the soil (in Colorado) 14 days after aerial application at 2.24 kg/ha. In a laboratory phase of the study, the half-life of the ester was 0.63 day, but there was no corresponding increase in the acid, suggesting hydrolysis was not the only process of ester disappearance. STARK and TORSTENSSON (1978) indicated 2,4-D butoxyethyl ester required at least 1 mon for hydrolysis in Swedish forest soils and that the "acid" form of herbicide persisted for periods ranging from 2 mon in central Sweden to more than 9 mon in the north. STEWART and GAUL (1977) reported the isooctyl esters of 2,4-D and 2,4,5-T were not detected in soil 13 hr after application of 1.1 to 4.3 kg/ha of herbicide. At rates of

Table VIII. *Half-lives of herbicide in 3 soils* (ALTOM and STRITZKE 1973).

	Quachita Highlands		Cross-timbers
Herbicide	Forest (days)	Grassland (days)	Forest (days)
2,4-D	5	4	4
Dichlorprop	12	8	10
2,4,5-T	24	14	21
Silvex	21	14	15

application of 31 kg/ha, a small percentage of the ester persisted for several wk in soil. The major soil residues resulting were the respective free phenoxy acids which in turn decreased with time. CHENEY *et al.* [not dated] looked for both acid and ester forms of 2,4,5-T after aerial application of 3.36 kg/ha of 2,4,5-T as PGBE ester. They reported a rapid decline in the level of ester in the soil (9.2 mg/kg immediately after spraying, 1.6 mg/kg 3 days later and <0.04 mg/kg in 32 days) but no concomitant increase in the acid form. These data suggest that what has been assumed by some investigators to be ester hydrolysis is in fact loss due to volatilization, photochemical degradation, or some other process.

In a field study, ALTOM and STRITZKE (1972) reported dichlorprop and 2,4-D were completely dissipated from soil beneath a blackjack oak and post oak forests in 30 days, while 2,4,5-T residues persisted from 60 to 90 days. STEWART and GAUL (1977) found the maximum soil residues of 2,4-D and 2,4,5-T acid occurred at 14 to 22 days after application of isooctyl ester. The length of persistence varied with the rate of application, and 2,4,5-T residues were more persistent than 2,4-D residues at all application rates. The residues of both herbicides were found primarily at depths of zero to ten cm. With both herbicides, more than 90% of the applied chemical had disappeared within 70 days, and after 55 wk, residue levels were 0.1% of the initial application or less. In no case were residues found deeper than 20 cm after 55 wks. In a limited monitoring study in Minnesota, 2,4-D residues in the zero- to 7.5-cm soil zone decreased about 90% in 31 days after aerial application (1 mg/kg at 3 days, 0.94 mg/kg at 18 days, and 0.11 mg/kg at 31 days after application) (*U.S. Department of Agriculture, Forest Service* 1978 a). There was no significant leaching of 2,4-D below 7.5 cm in the soil.

PLUMB *et al.* (1977) reported on the persistence characteristics of 2,4-D and 2,4,5-T applied at 4.48 kg/ha ae each in 186 L of water/ha to a chamise site in southern California. Residue levels were not measured immediately after application, but both 2,4-D and 2,4,5-T showed a half-life of about 19 days for the period 14 to 29 days after application. Clearly, the rate of degradation changed with time (Table IX). Approximately 1 yr after application, residues for both herbicides were 0.04 to 0.05 mg/kg.

Table IX. *Average concentration of herbicide in soil from a chamise site in southern California* (PLUMB *et al.* 1977).

Days after treatment	2,4-D (mg/kg) at soil depth (cm)			2,4,5-T (mg/kg) at soil depth (cm)		
	0–10	10–20	20–30	0–10	10–20	20–30
14	1.16	0.16	0.09	0.88	0.06	0.03
29	0.71	0.07	0.05	0.53	0.02	0.02
69	0.22	0.02	0.02	0.29	0.01	0.03
146	0.11	0.02	0.01	0.21	0.02	0.01
379	0.04	0.02	0.02	0.05	0.03	0.03

The relatively rapid initial dissipation of both herbicides in the zero- to ten-cm soil zone is somewhat surprising. The low organic matter conten the texture of the soil, and the extremely dry conditions encount d during the first 5 mon after application do not favor microbial act y which is most often reported to be the responsible agent for phenoxy herbicide dissipation. Volatilization and photodecomposition may have been important processes in residue reduction at this site. BOVEY and BAUR (1972) suggested herbicide loss can be partially attributed to photodegradation and volatilization. Herbicide residues detected by PLUMB *et al.* (1977) in soil zones below ten cm are believed to be due to contamination during sampling of the dry, loose soil in the upper soil zones. Lack of rainfall for the first 29 days after application precludes leaching for this period.

At a grassland study site in Texas, BOVEY and BAUR (1972) found a rapid reduction in 2,4,5-T residue levels with time after application. Average initial herbicide residues for 5 different sites receiving 1.12 kg/ha was 4.23 mg/kg in the zero- to 15-cm soil zone. Six weeks after application these residue levels had declined to 0.03 mg/kg. There was little or no leaching of 2,4,5-T at the 5 study sites despite 13 to 45 cm precipitation within the first 12 wk of application. SCIFRES *et al.* (1977) reported residues of 2,4,5-T in the zero- to 2.5-cm zone declined to trace levels 28 days after application of 2,4,5-T and picloram (1.12 kg/ha each) at 3 range sites in Texas.

NORRIS *et al.* (1977) determined 2,4,5-T residues in forest floor and soil after 2 successive annual applications (2.24 kg/ha ae isooctyl ester in diesel oil by helicopter in March). The study area was a cool, moist site in western Oregon (Table X). The rate of decline in 2,4,5-T levels in forest floor after the first application at this site was slower than at the hot, dry site in southern California (PLUMB *et al.* 1977), which may reflec the importance of volatilization and photodecomposition on the lo phenoxy herbicides from exposed soil surfaces. The rate of loss of 2,4, T was quite rapid the first 30 days after the second application, which indicates good adaptation of the microorganisms after the first application. One year after application, residue levels in forest floor were about 0.75% of the amount of herbicide originally applied.

In West Virginia, 2,4,5-T dissipated rapidly in the forest floor, despite substantial inputs in throughfall precipitation which occurred during the first 8 mon after application (NORRIS *et al.* 1978). At 1 wk, 52% of the level measured immediately after application still remained. At 3 mon 26% and at 12 mon 6% remained (Table X). These data show 2,4,5-T rapidly disappears from a variety of forest sites. There was some leaching of 2,4,5-T from the forest floor into the top 15 cm of soil, but movement into deeper soil zones did not occur (Table X) after the first application at either the Oregon or the West Virginia study sites. A similar pattern occurred after the second application at the Oregon site.

In Finland, ERONEN *et al.* (1979) found MCPA persisted for longer

Table X. *2,4,5-T in forest floor and soil after aerial application of 2.24 kg/ha 2,4,5-T ae in Oregon and West Virginia.*[a]

Residue location	Months after application 0	1	3	6	12	24
	First application—Oregon					
Forest floor (mg/m^2)	35.7	40.6	12.1	3.9	1.7	0.7
Soil (mg/kg)						
0–15 cm	0.01	0.02	0.08	0.02	0[b]	0
15–30 cm	0	0	0	0	0	0
30–45 cm	0	0	0	0	0	0
45–60 cm	0	0	0	0	0	0
	First application—West Virginia					
Forest floor (mg/m^2)	93.9	40.8	24.3	19.3	5.9	4.7
Soil (mg/kg)						
0–15 cm	0.34	0.03	0.03	0.02	0.03	0.01
15–30 cm	0	0.01	0	0	0	0
	Second application—Oregon					
Forest floor (mg/m^2)	137.4	9.7	12.5	4.1	1.5	—[c]
Soil (mg/kg)						
0–15 cm	0.01	0	0	0	0	—
15–30 cm	0	0	0	0	0	—
30–45 cm	0	0	0	0	0	—
45–60 cm	0	0	0	0	0	—

[a] Oregon data from Norris *et al.* (1977), West Virginia data from Norris *et al.* (1978).
[b] 0 means less than 0.01 mg/kg in soil or 0.1 mg/m^2 in forest floor.
[c] No samples were collected 24 mon after the second application.

periods in forest floor than has been reported for other phenoxy herbicides in the U.S. Unfortunately the data are expressed on a concentration basis (both fresh and dry wt) so allowances for variation in depth of forest floor among sampling locations is not possible. The values reported (dry-wt basis) are 112 mg/kg at 40 days, 95 mg/kg at 261 days, and 12 mg/kg at 283 days after application (2.5 kg/ha of MCPA isooctyl ester). In the soil which presumably was overlain by the forest floor, MCPA residues (dry-wt basis) were not detected immediately after application nor at depths greater than 3 cm more than 40 days after application. At 15 days, the residues were 4.0 mg/kg (0 to 3 cm), 0.3 mg/kg (3 to 6 cm), and 0.1 mg/kg (6 to 15 cm). Residues were as follows in the zero- to 3-cm zone of soil: 4.0 mg/kg at 15 days, 1.1 mg/kg at 40 days, 0.2 mg/kg at 283 days, and 0.7 mg/kg at 297 days.

In laboratory studies Norris (1970 b) found that the phenoxy herbicides were rapidly adsorbed by forest floor material, but the rate of desorption was also quite rapid, suggesting a low energy of adsorption and relatively good mobility in forest floor material with abundant

rainfall. In the field, on the other hand, most studies showed relatively little leaching of phenoxy herbicides. Data in Table X suggest less movement of 2,4,5-T into soil from forest floor after the second application at the Oregon site, despite the fact that initial herbicide residue levels in the forest floor are nearly 4 times higher. This is accounted for by the rapid reduction of residue levels in the forest floor after the second application (probably from good microbial adaptation for 2,4,5-T degradation in soil after the first application). The initial rate of disappearance of 2,4,5-T in soil in West Virginia was rapid, but the rate slowed appreciably after the first mon.

d) Behavior in water

Phenoxy herbicides can enter surface water by direct application to stream surface, accidental drift from nearby treatment units, mobilization (during wet periods) of herbicide deposited in dry, ephemeral stream channels, overland flow during periods of intense precipitation, or leaching through the soil profile. The magnitude and duration of surface water contamination which might occur (and therefore organism exposure) from each of these processes is different. Direct application or drift to surface waters is likely to result in the highest concentrations of herbicide in the water, but the duration of entry is short, being largely restricted to the period of application. Therefore, organism exposure may be relatively intense, but brief. Mobilization of residues in ephemeral stream channels will occur only during the first few periods of heavy precipitation after application. Residue levels may be initially high but will decrease rapidly as stream levels rise and as residues are washed away. Organism exposure may resemble that resulting from direct application or drift.

Leaching and surface runoff are two competing mechanisms that move chemicals from spray deposits to streams. Most rainfall reaching the ground either enters the soil or flows over its surface. In both cases, surface-deposited chemicals are carried along either in solution or adsorbed on suspended matter. The nature of the surface and of the precipitation are the factors which influence the distribution of water between surface flow and infiltration. More specifically, these include (1) amount of surface organic matter, (2) slope, (3) depth of soil profile, (4) infiltration characteristics of soil, (5) intensity, duration, and form of precipitation, and (6) immediate previous precipitation history. Factors favoring infiltration will decrease the amount of surface runoff and with it the overland flow of chemical.

The amount of chemical actually entering a stream due to surface flow will be influenced by (1) distance from stream course to closest point of chemical application, (2) infiltration properties of soil or surface organic matter, (3) rate of surface flow, and (4) adsorptive characteristics of surface materials. Conditions which retard the rate of surface runoff

will minimize the immediate level of stream concentration and also reduce the long-term stream load of pesticide, because a longer residue time in the soil provides greater opportunity for adsorption and degradation.

If precipitation is sufficient to cause overland flow, stream discharge volumes are likely to be considerably greater than during periods of application, and therefore herbicide concentrations would probably be lower. The duration of entry via overland flow would be brief, being restricted to periods of particularly intense precipitation. If herbicides enter streams by leaching, the concentrations will be quite low, but the duration of entry could conceivably be considerably longer than for either direct application or the overland flow process. The role of each of these processes in relation to stream contamination from operational use of phenoxy herbicides in forestry are considered in the following sections.

1. Entry to streams via leaching.—On a relative scale, phenoxy herbicides are considerably more mobile in the soil than many pesticide materials, but in fact their movement is still quite small compared to the distance from treated areas to streams (Harris 1967 and 1969). Numerous investigators have shown herbicide persistence and mobility in the soil are inversely correlated with organic matter. Laboratory or agriculturally based studies by O'Connor and Wierenga (1973), O'Conner and Anderson (1974), Edwards and Glass (1971), Lutz *et al.* (1973), Wiese and Davis (1964), Helling (1971 a, b, and c) and forest- and range-based studies by Norris *et al.* (1977), Plumb *et al.* (1977), and Bovey and Baur (1972) all support the hypothesis that leaching of phenoxy herbicides is not an important process for transporting significant quantities of these chemicals to streams.

2. Entry to streams via overland flow.—This process requires overland flow of water, a phenomenon hydrologists report is relatively uncommon on most forest sites. The infiltration capacity of the forest floor and soil far exceeds most rates of precipitation. Infiltration capacities in excess of 100 cm/hr are not uncommon in many forest environments. Overland flow may occur in areas where soils are badly compacted, are water repellant, or have no surface protection in the form of forest floor or vegetation.

Barnett *et al.* (1967) maximized the probability of runoff by applying artificial rain (6.25 cm/hr) to recently tilled agricultural land and found 38% of the 2,4-D isooctyl ester but only 5% of the 2,4-D amine salt in the washoff (mixture of eroded soil and water). The concentration of 2,4-D (isooctyl ester) in the washoff increased from 0.9 to 1.95 mg/L as the temperature of the artificial rain increased from 8° to 27°C (1 day after application). Equally, however, the concentration decreased most rapidly during the one-hr test rain when the rain was 27°C than when it was 8°C (estimated half-time was 18 min at 27°C and 65 min at 8°C).

Norris (1969) reported runoff of 2,4-D and picloram from a powerline right-of-way in southwestern Washington where there was evidence of soil compaction. The highest concentration of 2,4-D in runoff water (0.825

mg/L) was associated with the first significant storm after application. The concentration decreased with subsequent storm events even thoug those storms were larger than the first storm. No residues were found beyond about 6 wk after the first runoff event. TRICHELL *et al.* (1968), BOVEY *et al.* (1974), and SCIFRES *et al.* (1977) also reported research demonstrating runoff of herbicides from agricultural or rangeland situations.

Where runoff does occur, the concentration of herbicide in the transporting water may be reduced as it passes over unsprayed areas. ASMUSSEN *et al.* (1977) applied simulated rainfall 1 day after application of 0.56 kg/ha 2,4-D to plots planted with corn. The runoff from the plots was directed through a 24-m-long grassed waterway. Of the rainfall applied one day after application of herbicide, 50% ran off under dry antecedent soil conditions and 78% ran off under wet conditions. Reduction of suspended sediment in the waterway was 94 to 98% of the total amount of sediment moving from the plot. The total amount of 2,4-D moving from the plot was 2.5% and 10.3% in the dry and wet antecedent soil conditions. Of the 2,4-D lost in solution from the plot, about 70% was removed from solution as the runoff flowed over the 24-m grassed waterway. TRICHELL *et al.* (1968) also found substantial decrease in both the concentration and total discharge of picloram when the contaminated water ran over untreated soil.

Increased outflow of phenoxy herbicide from treated watersheds may occur with heavy rains, but the process is more likely to involve the mobilization of surface residues from streambanks and ephemeral stream channels rather than overland flow. Mobilization occurs as the stream network expands during precipitation. Research on this process is reported in the following section.

3. **Direct application or drift to surface waters and mobilization in ephemeral stream channels.**—Direct application and drift to surface water or ephemeral stream channels are the principal routes of phenoxy herbicide entry to forest streams, lakes, and ponds. These are also the processes over which the applicator and forest manager can have the greatest influence. Substantial research, operational, and regulatory experience has been gained in this area. PATRIC (1971) and NORRIS and MOORE (1971) provided useful compilations of some studies of herbicide entry to forest streams. The following paragraphs describe and discuss results of studies of research and monitoring for stream contamination in connection with the operational use of phenoxy herbicides in forest and range sites. Most of the residues detected in these studies resulted from direct application or drift to surface water or ephemeral stream channels. In those cases where monitoring continued for long periods, data related to leaching and overland flow are also included.

NORRIS (1967), reporting research done by Norris, Newton, Zavitkovski, and Freed, presented data on herbicide residues in streams from

several watersheds in Oregon forests treated with 2,4-D, 2,4,5-T, or a combination of the 2 herbicides. All treatments were low-volatile esters in diesel oil or water applied by helicopter at rates ranging from 1.12 to 3.36 kg/ha ae. The results showed some herbicide was present in nearly every stream which passed through, or was adjacent to, treated areas.

Maximum concentrations ranging from 0.001 mg/L to 0.13 mg/L occurred during or shortly after application. The highest concentrations and longest persistence occurred when no provisions were made to avoid direct application to stream surfaces in the treated areas. The time required to return to nondetectable levels ($<$0.001 mg/L) varied with the nature of the area treated and the maximum herbicide concentration observed. Times ranging from less than 1 hr to more than 168 hr have been noted, with less than 1 day required in most instances.

In Alaska, SEARS and MEEHAN (1971) found no residues in stream water the first 2 days after application of 2.24 kg/ha ae 2,4-D butyl ester. On the third day after application, however, one sample contained 0.2 mg/L 2,4-D. A 30-m unsprayed buffer strip was left along the main part of the stream to minimize entry of herbicide to the stream, but no effort was made to avoid application to numerous small tributaries. They also noted that wind currents during the spray operation caused some deposit in the main stream despite the buffer strip.

NORRIS (1967) found herbicide applications to marshy areas (areas with a high water table) can lead to higher than normal levels of stream contamination. In one instance, concentrations approaching 1.0 mg/L of 2,4-D were found in water flowing from a marshy area. 2,4-D continued to flow from the area in detectable concentrations for the duration of a 168-hr sampling period. He noted, "This particular situation (treating marshy areas) is probably one of the most dangerous in terms of potential stream contamination since a very slight rise in the water table could result in the release of very large quantities of herbicide to streams draining such an area."

ALDHOUS (1967) used a thin-layer chromatographic system (reported sensitivity of 0.005 mg/L) to monitor 2,4-D in surface water in the bottoms of ploughed furrows in a peat area treated with nonyl ester of 2,4-D (applied by aircraft at 4.5 kg in 155 L of water/ha). High levels of herbicide (1.5 to 2.0 mg/L) were found for the first 7 days, but no detectable residues were found after 28 days. The author indicated these residues were the result of drainage water mobilizing 2,4-D residues originally deposited in the furrow bottoms (the same mechanism which operates in ephemeral stream channels). Stream contamination could result if substantial surface drainage from this type of area occurs (BARNETT *et al.* 1967).

NORRIS (1967 and 1968) looked for the long-term entry of 2,4-D and 2,4,5-T into forest streams draining areas receiving these herbicides. In one study, 11 streams in western Oregon were monitored immediately

below treatment areas on a regular basis for 9 mon after application. Once the initial stream contamination had declined to nondetectable limits, herbicide residues were detected during first fall rains or subsequent periods of heavy precipitation. In a second study, two other watersheds in western Oregon were studied. In one, the treatment area bordered a stream for more than 3 km and extended 200 to 400 m upslope. 2,4-D and 2,4,5-T were applied at 1.12 kg/ha ae each as low-volatile esters in diesel oil in the spring. The second area contained 25 different treatment areas totalling 160 ha which received the same treatment in a 1135-ha watershed. In both cases, streams were sampled to detect the movement of herbicide from treated areas to the stream during the first fall storms which raised stream levels. No residues were found.

Norris, Newton, Zavitkovski, and Freed (see NORRIS 1967) noted a rapid decrease in herbicide concentration with downstream movement. An estuary receiving water from a large forest area which included numerous herbicide treatment areas showed no detectable phenoxy herbicides (<0.001 mg/L). Through the use of buffer strips and careful attention to details of application, phenoxy herbicide concentrations in forest streams will seldom exceed 0.01 mg/L and will not persist for more than 24 hr. Public and private forest management groups in the Pacific Northwest have monitored streams for herbicide residues during and after operational brush control operations since the mid-1960s. The results of these efforts confirm the findings discussed above.

In southern California, KRAMMES and WILLETS (1964) sampled for 2,4-D and 2,4,5-T in a small stream after application to control riparian vegetation. Maximum concentration detected was 0.09 mg/L and residues were found in only 2 samples; however, the sampling intensity was not great.

In an Arkansas forest, LAWSON (1976) sampled stream water during rising hydrograph to look for storm-induced herbicide runoff after the [illegible]ing two 0.6-ha watersheds with 4.48 kg/ha ae 2,4,5-T in diesel oil by backpack sprayer in 3 successive yr. The sampled streams were ephemeral, not perennial, and flowed only during significant storms. 2,4,5-T residues up to 2.2 mg/L were detected in water collected in connection with the first runoff event which occurred 17 days after application. Less than 0.2 mg/L of 2,4,5-T was detected in the perennial stream which received the ephemeral streamflow from this area. Barely detectable levels of 2,4,5-T were found in samples collected during the next runoff event, approximately 7 wk after application. Subsequent storms did not produce detectable 2,4,5-T residues. These residues appear to be the result of herbicide mobilization from stream channels which were dry at the time of treatment. No herbicide residues were detected in samples collected during runoff events after either the second or the third application. These latter results are difficult to interpret but may suggest decomposition of the herbicide by microbial populations adapted to the use of 2,4,5-T after the first application. In any case, it is clear that in this Arkansas forest sig-

nificant movement of 2,4,5-T from treated areas to perennial streams did not occur.

In a similar forest type in Oklahoma, IGLEHEART *et al.* (1974) measured 2,4,5-T residues in water collected from streams immediately below 4 areas treated at a rate of 2.24 kg/ha ae as 2,4,5-T PGBE ester applied by helicopter in May and June. Treated areas ranged in size from 100 to 800 ha and constituted 20 to 100% of the watershed. The results were similar to those of NORRIS (1967). Maximum concentrations occurred shortly after application, and residues declined to low or nondetectable levels in a short time (Table XI). Maximum herbicide concentrations were quite similar to those in Pacific Northwest streams which originated in or ran directly through treated areas and where no effort was made to avoid direct application to stream surface. IGLEHEART *et al.* (1974) noted that the first significant rains did not introduce important quantities of 2,4,5-T into the streams.

SUFFLING *et al.* (1974) monitored for herbicide runoff in an Ontario forest for 1 yr after application of a 1:4 mixture of picloram:2,4-D (9.35 kg/ha ae) to a powerline right-of-way. The first runoff event took place the day of application, and 0.001 mg/L of 2,4-D was detected. This is the only time detectable levels of 2,4-D occurred at this site which again suggests mobilization of residues in ephemeral stream channels as the probable mechanism rather than overland flow.

REIGNER *et al.* (1968) used odor tests to look for phenoxy herbicides in Pennsylvania and New Jersey streams in 4 forest areas treated with butoxyethanol or emulsifiable acid formulations of 2,4,5-T applied by mist blower. About 0.04 mg/L of herbicide was detected immediately after application, but residue levels declined about 50% in 4 hr, and no residues were detected in samples collected at various intervals over the next 4 wk. Samples collected from the Pennsylvania streams in connection with the first storm to produce more than 2.5 cm precipitation contained

Table XI. *Herbicide in stream water from Oklahoma forests treated with 2.24 kg/ha of 2,4,5-T by helicopter* (IGLEHEART *et al.* 1974).

Time after application	Residue (mg/L)[a]			
	Site 1	Site 2	Site 3	Site 4
Prespray	0	0	0	0
1 hr	0.027	0.001	0	0
4 hr	0.145	0.018	0.023	0.031
1 day	0.010	0.011	0.005	0.002
7 days	0	0	0.002	0
14 days	0	0	0	0
30 days	0	0.002	0	0
After first significant rain	0	0.001	0.001	0

[a] 0 means <0.001 mg/L.

0.01 and 0.02 mg of 2,4,5-T/L. The New Jersey streams did not contain detectable herbicide after a similar storm. This study is limited by the nonspecific detection method used.

In West Virginia NORRIS *et al.* (1978) studied the fate of 2,4,5-T in a 22-ha watershed aerially sprayed at 2.24 kg/ha ae of butoxyethanol ester. A maximum of 0.05 ppm of 2,4,5-T was found in stream water collected immediately below the treated areas about 3 hr after application (following a brief but intense rain shower of 1.3 cm the first hr after application) (Fig. 2). Residues were detected sporadically the first 11 days after application, with highest concentrations occurring during 3 rainy periods. No residues greater than 0.01 mg/L were detected more than 1 day after application. No residues (<0.001 mg/L) were found more than 11 days after application, although sampling continued for more than 2 yr. About 0.0175% of the 48 kg 2,4,5-T applied to this watershed was discharged in streamflow (NORRIS 1980). No residues were found at 2 downstream locations.

PIERCE (1969) applied 2,4,5-T and 5-bromo-3-*sec*-butyl-6-methyluracil (bromacil) by backpack sprayer to prevent revegetation on an experimental watershed in New Hampshire. Samples were collected for more than 1.5 yr, and the concentration of 2,4,5-T did not exceed 0.001 mg/L at any point although during this same period the concentration of bromacil was as high as 1.4 mg/L. The bromacil had been applied at 28 kg/ha by helicopter and, although the data are limited, it appears 20% of the bromacil left the watershed in streamflow within the first 1.5 yr after treatment.

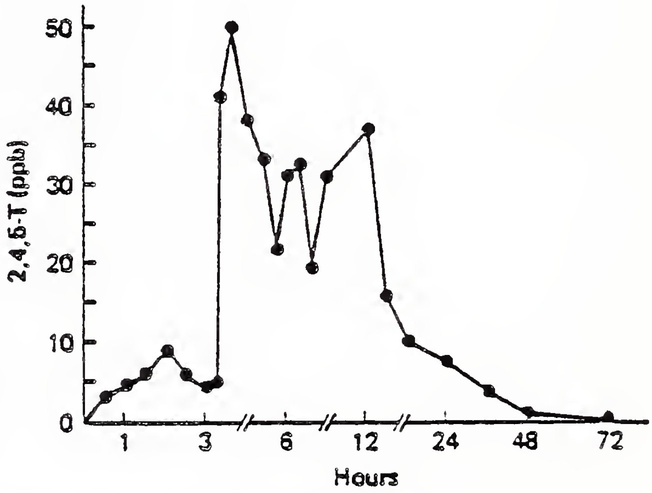

Fig. 2. Concentration of 2,4,5-T in a forest stream immediately downstream from a 22-ha watershed aerially sprayed with 2.24 kg/ha ae of butoxyethanol ester of 2,4,5-T, West Virginia (NORRIS *et al.* 1978).

In North Carolina, DOUGLASS *et al.* (1969) applied 2,4-D and 2-chloro-4-ethylamino-6-isopropylamino-*s*-triazine (atrazine) with groundspray equipment to convert a hardwood stand to grass in an effort to increase water yield. In the first application only atrazine was applied and no buffer strip was left. Atrazine to 0.04 mg/L was detected in the stream with highest concentrations occurring during rainstorms a few days after application. In the second application a 3-m buffer strip was left between the stream channel and the area receiving 2,4-D (isobutyl ester at 3.36 kg/ha ae). No 2,4-D was found in the stream draining the 8-ha watershed even during several storms depositing more than 2 cm of rain/day in the first 2 mon after the second application. EDLUND and TANGDEN (1977) reported only one sample out of 18 taken at 6 sites on 3 occasions showed any trace of phenoxy herbicide (<0.005 mg/L) after spraying (aerially) a test area (in Sweden) with MCPA or 2,4-D.

Limited scale monitoring of operational applications of phenoxy herbicides on forest lands is largely in agreement with the research reports cited in this section. Various reports of some of these efforts contain some data on residues of 2,4-D, silvex, and 2,4,5-T in water (*U.S. Department of Agriculture, Forest Service* 1978 a, b, and c). *U.S. Department of Agriculture, Forest Service* (1980) recently analyzed reports of the results of monitoring for phenoxy herbicides in streams after operational application in forests of the northwestern United States. The reports analyzed represented 680 samples from 304 different applications. In most cases, a composite sample was made from 2 to 5 individual samples. They found 84% of the samples did not contain herbicide residues, 14% had residues up to 0.005 mg/L, 1% had between 0.005 and 0.01 mg/L, and 1% contained more than 0.01 mg/L.

HAMMERSTROM *et al.* (1971) reported the results of a "one time sampling" survey of 58 community water supplies for 2,4-D, 2,4,5-T, and silvex. 2,4,5-T was detected in 11 or 19% of the water supplies at concentrations ranging from <0.0005 to 0.00057 mg/L. 2,4-D at levels of <0.0005 to 0.0034 mg/L was detected in 31% of the water supplies. Silvex was detected in 6.9% of the water supplies at concentrations of <0.0005 mg/L. Most of the water supplies sampled were downstream from agricultural or mixed agricultural and forestry areas. Most samples reported as positive for phenoxy herbicide were less than the quantitative level of detection (0.0005 mg/L). Of the 6 samples that contained more than 0.001 mg/L (all 2,4-D, no 2,4,5-T or silvex), 5 were from Oklahoma.

BROWN and NISHIOKA (1967), MANIGOLD and SCHULZE (1969), and SCHULZE *et al.* (1973) surveyed western U.S. streams for pesticides between 1965 and 1971. No phenoxy herbicides were found in any of the streams sampled by BROWN and NISHIOKA (1967), but 2,4-D, 2,4,5-T, and silvex were detected in many streams by MANIGOLD and SCHULZE (1969) and in every stream sampled (but not every sample collected) by SCHULZE *et al.* (1973). The maximum concentration detected was 0.00099

mg/L of 2,4-D. There was substantial variation from yr to yr and among locations. In nearly every case the sampled streams were in or receive runoff from nearby agricultural lands.

These various studies and monitoring programs largely support the conclusion that direct application and drift to surface water and mobilization in ephemeral stream channels are the principal sources of phenoxy herbicides in forest waters. Overland flow and leaching do not appear to be important. Residues in water from direct application and drift can largely be controlled through careful orientation of spray units to streams and by careful attention to climatic, equipment, and application factors.

4. **Fate in streams.**—The fate of phenoxy herbicides in forest stream systems has received little direct study, although some inferences can be made from monitoring studies. NORRIS and MOORE (1971) presented some information on this point, but their inferences are largely based on studies done in the laboratory or in surface waters not particularly associated with forests. PARIS and LEWIS (1973) summarized the work reported in the literature between 1945 and 1971 concerning the photochemical, chemical, and microbial degradation of 10 selected pesticides, including 2,4-D butoxyethyl ester, in water. The processes which operate on the phenoxy herbicides in the terrestrial environment also operate in aquatic systems, but the rates are likely to be different.

Adsorption, uptake by organisms, volatilization, and downstream transport are processes which, while they do not eliminate phenoxy herbicides from the environment, reduce the exposure organisms in a particular area may receive. Degradation—microbial, chemical, or photochemical—is the only process which can reduce total environmental load. Widely divergent results have been obtained in studies of phenoxy herbicide behavior in the aquatic environment. Some of this divergence results because persistence reflects the combined result of numerous processes which change the level of herbicide in one area, and the rate at which one process operates varies widely among areas.

a) Hydrolysis of phenoxy herbicide esters.—Phenoxy herbicide esters hydrolize in water at pH, temperature, and buffer conditions which are found in nature. The esters are more toxic than the acid or salt forms, thus rapid hydrolysis will substantially reduce the potential for toxic impacts on aquatic organisms. The half-lives for hydrolysis of various phenoxy esters varied from 4.5 to more than 88 days at pH 7.4 (Table XII); however, persistence of the acid forms (the products of hydrolysis) may be lengthy (STRUIF *et al.* 1975).

KENAGA (1974) (reporting the unpublished research of Teasley and Williams) showed that at concentrations of about 1 mg/L ae the following % hydrolysis of 2-ethylhexyl ester of 2,4,5-T occurred: 42% in 4 hr, 67% in 8 hr, and 88% in 16 hr. Ultraviolet light was an important factor in the hydrolysis of esters of both 2,4,5-T and silvex. ZEPP *et al.* (1975) used theoretical considerations (and some data) to calculate the half-lives of a series of 2,4-D esters. At pH 9.0, the half-lives were: methyl (1.1 hr),

Table XII. *Hydrolytic half-lives of phenoxyacetic acid esters in water at pH 7.4.*[a]

Chemical	Phenoxyester	Half-life (days) at 4°C	Half-life (days) at 20°C
2,4-D	-methyl	44.7	4.5
	-*n*-butyl	—	5.0
	-isopropyl	—	23.1
2,4,5-T	-methyl	36.4	—
	-*n*-butyl	48.5	7.5
	-isobutyl	61.0	10.2
	-*n*-amyl	57.4	8.4
	-3-methylbutyl	88.2	11.6

[a] From Table 3 of Struif *et al.* (1975).

isopropyl (17.0 hr), *n*-butyl (5.2 hr), *n*-octyl (5.2 hr), and isooctyl (37.0 hr). At pH 6.0, the half-lives were longer: methyl (44 days), isopropyl (710 days), *n*-butyl and *n*-octyl (220 days), and isooctyl (1,500 days). Clearly, both biotic and photolytic factors are important in ester hydrolysis because persistence at near-neutral pH of the length calculated by Zepp *et al.* (1975) does not seem to occur in natural conditions.

Aly and Faust (1964) found 2,4-D esters were hydrolyzed to the acid form in 9 days in lake water. Rodgers and Stalling (1972) reported a half-life of about 24 hr for the BEE ester of 2,4-D. In a field test where 2,4,5-T (PGBE ester) was applied at 3.36 kg/ha, Cheney *et al.* [not dated] reported the rapid hydrolysis of ester in the water of a trough that was directly sprayed. 2,4,5-T ester was highest (0.364 mg/L) 1 day after application; 2 days later no ester ($<$0.001 3 mg/L) was detected. The level of 2,4,5-T acid, however, had increased to a maximum level of 0.249 mg/L.

Cochrane *et al.* (1967) reported the propylene glycol butyl ether ester of silvex was rapidly and almost totally hydrolyzed to silvex acid in about 2 wk when applied at about 9 kg/ha to water over 3 different loam, sand, or muck soils. Between 1 and 7 days after application, the concentration of ester decreased from about 0.3 to 0.01 mg/L, while the silvex acid levels increased from about 0.4 to 1.6 mg/L. Less than 0.001 mg/L of ester remained at the end of two wks.

Bailey *et al.* (1970) reported 50% hydrolysis of the PGBE ester of silvex in treated ponds in 5 to 8 hr, 90% in 16 to 24 hr, and 99% in 33 to 49 hr. Both silvex ester and acid forms were found in the sediment, although surprisingly the ratio of acid to ester was about 10 in the sediment but about 0.1 in the water 4 hr after treatment. The ester had essentially disappeared in 5 wks.

β) *Adsorption of phenoxy herbicides.*—Phenoxy herbicide esters adsorb extensively to flocculated humic acids with the degree and strength of adsorption increasing with the number of carbon atoms in the alcohol

moiety. At neutral pH and with slight humic acid content, STRUIF *et al.* (1975) reported phenoxy acids were not appreciably adsorbed, but the adsorption of various esters was extensive. A nearly linear, inverse relationship existed between the water solubility and the Freundlich adsorption equilibrium constant. In natural water systems, both adsorption and hydrolysis will occur at the same time. STRUIF *et al.* (1975) reported the rates of hydrolysis of several phenoxy herbicide esters were about twice as fast in aqueous systems that also contained humic acid. This indicates hydrolysis may be more rapid when the ester is adsorbed, a hypothesis consistent with the data of BAILEY *et al.* (1970) who reported a ratio of silvex acid to ester of 10 in pond sediments 4 hr after application of silvex ester.

COCHRANE *et al.* (1967) reported both the ester and acid forms of silvex were adsorbed by the hydrosoil, although the authors were uncertain that the ester per se was being adsorbed. In followup laboratory tests, they found rapid equilibration of silvex ester and acid between the water and sediment phases. After 16 hr of shaking, more than 90% of the ester was adsorbed (either as acid or ester) on 2 of the 3 soils tested. The acid was also extensively adsorbed (86 to 100%) in the same period.

BAILEY *et al.* (1970) reported both the acid and ester forms of silvex were found in sediment, but the adsorption of the ester was not as extensive as would have been predicted from the data of STRUIF *et al.* (1975). BAILEY *et al.* (1970) indicated the adsorptive capacity of the pond sediments was saturated when the concentration of herbicide in water exceeded 2 mg/L. The apparent limited adsorption capacity may be due to the limited sediment-water interface. In flowing streams with some suspended sediment component in the water column, more extensive adsorption may occur.

Aquatic plants will also absorb some phenoxy ester from water. SMITH and ISOM (1967) reported 8.26 mg/kg of 2,4-D (butoxyethanol ester) in eurasian watermilfoil (*Myriophyllum spicatum* L.) 24 hr after application of the granular herbicide at 112 kg/ha ae. The water concentration was 0.037 mg/L 8 hr after treatment and <0.001 mg/L at 24 hr. Residues of 2,4-D in sediment ranged from 0.14 to 56.8 mg/kg. SCHULTZ and WHITNEY (1974) studied the distribution and fate of 2,4-D (4.48 kg/ha as the dodecyl-tetradecyl amine salt) in a canal in Florida. The highest concentration of 2,4-D in the water was 0.037 mg/L the day after treatment; no sediment sample contained more than 0.005 mg/kg (detection limit) 2,4-D.

Clearly, phenoxy herbicides, either as acids, salts, or esters, will interact with stream sediments. The net effect of this interaction is to reduce the instantaneous concentration of the chemical in the water. The total length of time the herbicide will be in the water may be increased as desorption from sediments occurs; however, the total load of aquatic pollutant will be reduced as the biological and chemical processes of

herbicide degradation occur while the herbicide is in the sediment. It is important, from the standpoint of evaluating toxic hazards to aquatic organisms and downstream water users to recognize and take into account the adsorption-desorption equilibrium which will develop between the water and sediment phases for all phenoxy herbicides in the aquatic environment (PIONKE and CHESTERS 1973).

γ) *Persistence of phenoxy herbicides.*—Phenoxy herbicides are also degraded (or disappear) from aquatic systems. SIKKA and OU (1976) and OU and SIKKA (1977) incubated silvex (300 mg/L) in a mixed culture of aquatic microorganisms developed by an enrichment technique from pond water and sediment. After 80 hr, essentially all the chlorine in the herbicide had been liberated as free chloride. Liberation of $^{14}CO_2$ from the ring-labeled silvex and loss of UV absorbance indicated ring cleavage had occurred. About 1 to 5% of the initial silvex was identified as trichlorophenol, which was also rapidly metabolized.

ROBSON (1966) reported 2,4-D degradad from 4.5 to <0.05 mg/L in fewer than 14 days in stagnant pond water. ROBSON (1968) also reported a marked decrease in 2,4-D persistence in fresh water (from 9 wks to 1 wk) when small quantities of soil previously treated with 2,4-D were added to the water. Presumably the soil contained its natural complement of microorganisms.

SONDERQUIST and CROSBY (1975) reported MCPA essentially disappeared from the water and sediment of a rice pond in California 14 days after application of 0.98 kg/ha of MCPA dimethylamine salt. Tests in other environments indicate persistence can be appreciable. SCHMIDT (1975) reported 100% of 2,4,5-T, 91% of MCPA, and 85% of 2,4-D were recovered 168 hr after their addition to water. ALY and FAUST (1964) reported 2,4-D persisted up to 120 days in lake water. In another study only 40% degradation of 2,4-D was observed in water in 6 mon under conditions that seemed excellent for biological activity (SCHWARTZ 1967). CHENEY *et al.* [no date] found highest residues of 2,4,5-T acid (0.249 mg/L) 3 days after aerial application of 2,4,5-T ester over a watering trough. The level of acid decreased erratically to 0.132 mg/L after 1 mon and 0.002 mg/L after 8.5 mon. In a pond treated with the BEE ester of 2,4-D, herbicide was no longer detected in water 36 days—and in hydrosoil 85 days—after treatment with 20% granules applied to achieve 1.33 mg/L of herbicide (FRANK and COMES 1967). COCHRANE *et al.* (1967) found rapid disappearance of silvex ester but longer persistence of silvex acid in water over sand, loam, or muck soils. Silvex acid decreased from 1.6 mg/L to about 0.05 mg/L in 18 wks.

DEMARCO *et al.* (1967) demonstrated the effect of temperature and level of oxygen in water on persistence of 2,4-D. They reported a considerable decrease in the rate of 2,4-D loss in biologically active natural river water at reduced levels of dissolved oxygen (Fig. 3). In a laboratory study CROSBY and WONG (1973) found the photodecomposition of 2,4-D

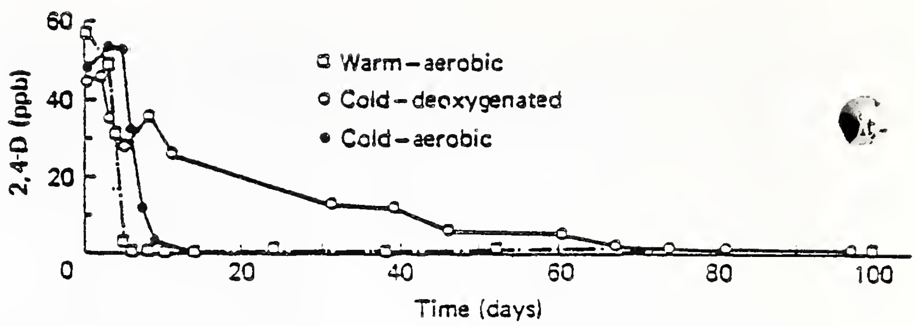

Fig. 3. The effect of temperature and oxygen level on persistence of 2,4-D in water (DEMARCO *et al.* 1967).

and 2,4,5-T was slow (half-life more than 1,800 hr). The authors felt photodecomposition may be more rapid in natural water. Rapid degradation of 2,4-D occured in water samples collected from areas with a history of repeated 2,4-D applications (GOERLITZ and LAMAR 1967). Some of these results may indicate some surface waters lack conditions suitable for biological activity or they may not contain populations of microbes adapted to the use of the phenoxy herbicides as substrates (HEMMETT and FAUST 1969).

AVERITT and GANGSTAD (1976) studied the rate of dissipation of 2,4-D at the point of application to waters for aquatic vegetation control. The results showed the rate of dissipation depended on the dosage rate, depth of water, mean temperature, and time since treatment. For application to the water surface of 4.48 kg/ha, there was a decrease of 0.058 mg/L for each 0.61-m increase in depth of water treated, 0.115 mg/L for each 7°C increase in temperature, and 0.053 mg/L for each 7 days after treatment.

The wide range of results presented in this section indicate the persistence of phenoxy herbicides in aquatic systems is influenced by many factors. The influences of environmental factors on the rates of clearance of 2,4-D and its derivatives from water are summarized in Table XIII.

5. **Stream loading.**—Most of the research on herbicides in the aquatic environment have only measured the concentration of chemical in the water. Without measurements of stream discharge for the same time period, it is not possible to estimate stream loading with any accuracy. NORRIS *et al.* (1976 c and 1981) have reported one of the few such studies done in the forest. They measured the discharge of herbicide in stream water from a 7-ha watershed in southwestern Oregon after aerial application of 2.3 kg/ha of picloram and 4.6 kg/ha of 2,4-D in June. The stream was intermittent and flowed only between November and May and during periods of heavy precipitation in late spring and early fall. About 0.022 ppm of 2,4-D was detected in the first outflow from the watershed

which occurred in October, about 3.5 mon after application. The last detectable level of 2,4-D (0.003 ppm) was found in samples collected 12 days later. Despite heavy rains and high rates of stream-water discharge from this watershed, no further outflow of 2,4-D was detected in samplings made over the next 2 yr. Of the 32.2 kg 2,4-D applied to the watershed, 4.5 g (0.014%) was discharged in stream water. The intermittent stream channel occupied 0.21% of the area of the watershed which suggests that residues detected represented mobilization of residues in the channel rather than either overland or subsurface flow. In West Virginia, NORRIS (1980) found 0.0175% of the 2,4,5-T applied to a 22-ha watershed was discharged in streamflow. All the discharge occurred in the first 11 days after application, although sampling continued for more than 2 yr.

FRANK and RIPLEY (1977) and FRANK *et al.* (1978) reported on the stream loading of 2,4-D, 2,4,5-T, and MCPA in 11 small agricultural watersheds in Ontario, Canada (Table XIV). While this is hardly the "forest environment," this study is unique in its scope, and the data are consistent with those reported by NORRIS *et al.* (1981) for 2,4-D and picloram and NORRIS *et al.* (1976 b) for trichlopyr.

SHEETS *et al.* (1972) reported on herbicide residues in surface runoff and flume water at the base of 2 rangeland watersheds (average declivity 35 to 40%) following treatment with picloram, 2,4-D, and 2,4,5-T at 2.24 and 4.48 kg/ha in 1967 to 1970. Maximum amounts moved in surface water over an 8-mon period after application in August 1968 were 0.049 and 0.015% for 2,4-D and 2,4,5-T. LANE *et al.* (1977) also measured herbicide discharge at the watershed level in a similar rangeland environment. They applied 2,4-D, 2,4,5-T, picloram, and 3,6-dichloro-*o*-anisic acid (dicamba) in 4 different zones of increasing proximity to the stream channel to evaluate the contribution of runoff from different parts of the watershed to the total watershed runoff. If the data for all herbicides is combined, 0.21 to 0.56% of the applied herbicide was recovered in streamflow in 78 days after application. Discharge was greatest with the first runoff event of any size (Table XV). No 2,4-D was found in any samples of stream water, which is consistent with its application zone being furthest from the stream (80 to 90 m). The concentration of 2,4,5-T did not exceed 0.025 mg/L. The authors did not give data for discharge of specific herbicides, but inspection of their data on concentration makes it clear little 2,4,5-T was discharged. The relatively persistent and mobile dicamba and picloram which were applied closest to the stream were the major components of herbicide discharge.

Although limited in scope, these data (along with data on herbicide levels in stream water) suggest stream loading will be low from application of phenoxy herbicides on forest and rangelands. More data is needed in this area, but the findings available thus far are consistent with the short persistence-limited mobility characteristics of phenoxy herbicides in soil.

Table XIII. *Effects of environmental factors on the rates of loss of 2,4-D from water.*[a]

Process	Temperature	Oxygen concentration	pH	Light intensity	Formulation	Other factors	References
Ester hydrolysis	Minimized by lowered temperatures For Butoxyethyl ester[c] $\ln k_b = \frac{-20.1}{RT} + \frac{14.8}{R} + \ln\frac{kT}{h}$ $\ln k_a = \frac{-17.6}{RT} + \frac{21.3}{R} + \ln\frac{kT}{h}$ at 0°C $k_b = 1\ sec^{-1}mole^{-1}$ $k_a = 2.8 \times 10^{-6} sec^{-1}mole^{-1}$	No effect	$t½ = \frac{0.693}{k_a[H^+] + k_b[OH^-]}$ For the butoxyethyl ester hydrolysis rate is minimized at a pH around 4. At pHs $\gtrsim 5$ $t½ = \frac{0.693}{k_b[OH^-]}$ pHs ≤ 4 relatively slow	n.s.[b]	See Section 2.1.1 and references in *National Research Council Canada* (1978) for details relating to particular ester formulations.	For more details see Section 2.1.1 in *National Research Council Canada* (1978). In sediments, the hydrolysis rates could be faster than those reported for aqueous solutions. Information is not available to illustrate this point. Since partitioning into the sediments will be minimized at pHs >4, this factor may be of little importance.	Zepp *et al.* (1975)
Volatilization	Minimum as temperature decreases	No effect	Volatilization will be negligible for the acid and its salts at pHs > 4. pH should have little effect on the rates for the ester formulations prior to their hydrolysis	No effect	Significant losses could occur prior to mixing or in shallow ecosystems in the case of the more volatile ester formulations. For details see Section 2.2.1 of *National Research Council Canada* (1978) and references.		Zepp *et al.* (1975) Mackay & Wolkoff (1973)
Microbial degradation	Pronounced effect not noted under aerobic or anaerobic conditions (slight reduction of rate in cold vs. warm aerobic conditions)	Aerobic conditions required for significant microbial degradation. Under anaerobic conditions, the half-life can exceed 80–120 days. Under aerobic conditions [illegible] half-life ca[illegible] short, in the o[illegible] of 1 to several	n.s.[b]	n.s.[b]	n.s.[b]	Application of 2,4-D in previous seasons may increase rate. Conditions favoring partitioning into the sediments may increase rate under aerobic conditions—i.e., formulation, pH, sediment, organic content must be considered.	Demarco *et al.* (1967) Schwartz (1967) Aly & Faust (1964)

Photolysis	Independent of temperature	Independent of oxygen concentration	n.r.[b]	Directly related to the intensity of radiation in the 280–320 nm range.	n.r.[b]	The significance of photolysis reactions is minimized during the winter months and in northern latitudes. Losses due to photolysis will be minimized in well-mixed deeper ecosystems.	Zepp *et al.* (1975)
Sorption	n.r.[b]	No effect	Only the ester and the molecular form of the acid are expected to partition to a significant extent into the sediments. pH will have little effect on esters prior to hydrolysis but pH < 3–4 required for other formulations.	n.r.[b]	See Section 2.2.2 of *National Research Council Canada* (1978) and pH.	The sediment partitioning should be related to the organic content of the sediments and, although not demonstrated, perhaps the nature of the organic material	Pionke & Chesters (1973)

[a] From Table 2.7, *National Research Council Canada* (1978).
[b] n.r. = not reported.
[c] k_a = acid hydrolysis, k_b = base hydrolysis,
$\frac{k}{h} = \frac{\text{Boltzman constant}}{\text{Planck's constant}}$, and t = temperature in °K.

Table XIV. *Stream loading of phenoxy herbicides on 11 agricultural mini-watersheds in Ontario, 1975.*[a]

Watershed	Vol of water ($x10\ m^3$)	Period	Herbicide	Loading (g)	Herbicide applied (%)	Months (%)
Big Creek	121	Mid Mar.–Dec.	2,4-D	60.5	0.014	May (0.2); June (89.1); July (3.9); Sept. (6.8)
			2,4,5-T	0.0	0.011	May (0.8); June (99.2)
Hillman Creek	65.5	Mid Mar.–mid Jan.	2,4-D	328	0.62	June (0.1); July (99.8); Nov. (0.1)
			MCPA	0.3		June (100)
Venison Creek	521	Feb.–Dec.	2,4-D	2.3	0.00037	July (100)
Canagagigue Creek	121	Mar.–Dec.	2,4-D	4.5	0.0026	June (43.1); July (0.1); Oct (0.4); Nov. (56.4)
			MCPA	2.7	0.0013	June (43.2); Oct. (0.4); Nov (56.4)
Little Ausable River	213	Early Apr.–early Jan.	2,4-D	2.8	0.00052	May (13.4); June (46.9); July (39.7)
Tributary of Maitland River	109	Late July–Dec.	2,4-D	3.0	0.00099	Oct. (100)
North Creek	79.5	Mid Apr.–mid Jan.	2,4-D	2.0	0.0035	June (100)
			2,4,5-T	2.0	—	June (100)
Tributary of Middle Thames River	88.9	Mid Apr.–mid Jan.	2,4-D	0.8	0.00041	June (49.4); July (50.6)
Shelter Valley Creek	460	Apr.–Dec.	2,4-D	4.4	0.0015	July (100)
			2,4,5-T	3.4		July (100)

[a] From Table 2.10, *National Research Council Canada* (1978); Frank and Ripley (1977); Frank *et al.* (1978).

Table XV. *Summary of runoff and herbicide yields from Watersheds 76.001 and 76.002 during the 1976 study.*[a]

Date of event	Days since treatment	Vol of runoff Event (L × 10^4)	Vol of runoff Cumulative (L × 10^4)	Yield of herbicides[b] Event (g)	Yield of herbicides[b] Cumulative (g)
		Watershed 76.001			
7/17/76	8	1.99	1.99	2.04	2.04
7/21/76	12	0.65	2.64	0.13	2.17
7/27/76	18	7.06	9.70	0.69	2.86
7/28/76	19	0.07	9.77	0.01	2.87
8/10/76	32	0.42	10.19	0.06	2.93
8/26/76	48	3.19	13.38	0.19	3.12
9/1/76	54	0.61	13.99	0.05	3.17
9/22/76	75	0.07	14.06	0.01	3.18
9/25/76	78	2.81	16.87	0.46	3.64
		Watershed 76.002			
7/17/76	8	0.69	0.69	0.95	0.95
7/21/76	12	0.03	0.72	0.05	1.00
7/27/76	18	7.36	8.08	8.41	9.41
8/26/76	48	5.49	13.57	0.53	9.94
9/25/76	78	2.40	15.97	0.64	10.58

[a] From Table 4 of Lane *et al.* (1977).
[b] No 2,4-D from Zone 1 was found in any of the water quality samples.

e) Bioaccumulation of phenoxy herbicides

Bioaccumulation is the uptake (and at least temporary storage) by an exposed animal of a chemical from the environment. Generally, bioaccumulation is more likely to occur when organisms are exposed to persistent chemicals of low water solubility and high lipid solubility. The phenoxy herbicides do not meet any of these criteria to the degree of the chlorinated hydrocarbon insecticides. Organisms exposed to a phenoxy herbicide, however, will take up some of the chemical. Generally, the bioaccumulation ratios will be low and the residence time brief once exposure ceases.

The fate of phenoxy herbicides in terrestrial animals (usually laboratory or farm animals) has received considerable study. These studies show rapid excretion of these herbicides and little or no residue detectable in fat (or milk) more than a few days after exposure ceases (Bjerke *et al.* 1972, Clark *et al.* 1975). Typically, more than 90% of phenoxy herbicide ingested is excreted unchanged in urine in 72 hr (Piper *et al.* 1973 b, Fang *et al.* 1973, Erne 1966).

Newton and Norris (1968) analyzed for 2,4-D and 2,4,5-T in a variety of tissues from blacktail deer collected in forest areas 15, 31, and 43 days after aerial application of herbicides. Measurable residues were found in at least one tissue from each animal collected. The highest concentra-

tions occurred in stomach contents (0.36 mg/kg), feces (0.16 mg/kg), urine (0.19 mg/L), and in one animal, the thyroid (0.15 mg/kg). Concentrations in tissues normally consumed by human beings were <0.006 mg/kg in heart, and ranged from <0.006 to 0.017 mg/kg in kidney, <0.006 to 0.021 mg/kg in liver, and <0.006 to 0.010 mg/kg in muscle.

In a Minnesota forest, a wide variety of birds and animals (31 in total) was trapped 36 to 266 hours after application of 2,4-D. About 59% contained detectable residues of 2,4-D with the average residue of 1.97 mg/kg (range <0.1 to 25 mg/kg). Average residue levels declined with time after application (6.93 mg/kg at 1 day, 0.82 mg/kg at 2 days, and 0.36 mg/kg at 14 days after application) (*U.S. Department of Agriculture, Forest Service* 1977).

As part of a widespread survey of the Swedish environment for phenoxy herbicide residues, ERNE (1974 and 1975) reported only 3% of 330 samples of muscle from healthy fish (several species from 120 locations) contained detectable residues of 2,4-D or MCPA. Residues ranged from 0.05 mg/kg to 1.5 mg/kg. Of 250 samples from wildlife found dead (numerous species including grouse, hare, deer, moose, and others), 25% of the samples of liver and kidney contained detectable residues, mostly 2,4-D and/or 2,4,5-T. The residue levels in liver and kidney were fairly low, the maximum value was 6 mg/kg in a roe deer kidney. Most of the values were <1 mg/kg. In 130 samples of liver and kidney from healthy moose, roe deer, and hare (shot in herbicide-treated forest areas), phenoxy residues were found in 42 animals (32%). Residues were all <4.5 mg/kg and most were <1 mg/kg. There was no marked difference in the pattern of residues between the group of animals found dead and those that were shot. However, only 2 of 60 (3%) of animals shot in untreated forest areas contained detectable residues.

GARCIA and RHODES (1979) reported residues of 2,4,5-T in muscle tissue of about 50% of the coots (*Fulica americana*) they collected around a lake which received runoff from Texas rangeland areas sprayed with 2,4,5-T. The average residue level detected was 0.2 mg/kg. A few birds had residues which averaged 0.02 mg/kg in fat, 0.04 mg/kg in liver, and 0.02 mg/kg in the gizzard.

In the aquatic environment, more work has been done on bioaccumulation, and generally uptake is limited and residence time brief. In an area treated with 1.12 kg/ha of 2,4-D ester (granular), mussels contained 0.38 and 0.70 mg/kg reflecting their consumption of algae which had absorbed the herbicide. Fish, on the other hand, did not contain detectable residues (<0.14 mg/kg) (SMITH and ISOM 1967). SCHULTZ and WHITNEY (1974) report 2,4-D residues in a variety of fish species ranged from undetectable to 0.162 mg/kg; about 80% did not contain detectable residues after application of 4.48 kg/ha of 2,4-D amine salt ae. SIGMON (1979) reported no measurable (<0.05 mg/kg) residues of 2,4-D in bluegill sunfish after 8 days in water containing 3 mg/L of 2,4-D as the butoxyethanol ester. 2,4,5-T residues ranged from 0.06 to 0.12 mg/kg under similar conditions.

YOCKIM *et al.* (1978) did an extensive bioaccumulation study with 2,4,5-T in a model aquatic ecosystem where the initial concentrations of 2,4,5-T in sediment were 0.1, 1.0, and 10 mg/kg (corresponding levels in water were 1.8, 15.2, and 143.2 mg/L 32 days later). Bioaccumulation ratios were highest in algae (about 50) and lowest in snails (about 5). Residue levels declined rapidly when the organisms were placed in fresh water for 7 to 14 days (Table XVI). VIRTANEN *et al.* (1979) studied the fate of MCPA and 4,6-dichloro-*o*-cresol (a 0.5%-level impurity of MCPA) in a model terrestrial ecosystem (Table XVII). They found rapid reduction of MCPA levels in terrestrial vegetation and some movement of MCPA to the water but no uptake of MCPA by aquatic plants. Both fish and snails contained detectable residue of MCPA but at levels less than the level in water, indicating no (or only a low) tendency for bioaccumulation. No residues of 4,6-dichloro-*o*-cresol were detected in any part of the system which authors attribute to its rapid degradation to the corresponding catechol.

IV. The behavior of TCDD in the forest

Much of the controversy about the use of 2,4,5-T and silvex in forestry involves 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). TCDD is produced as a by-product during one of the manufacturing steps of these herbicides. Although it is present in only minute quantities [EPA guidelines are <0.1 mg/kg; current production grade 2,4,5-T contains <0.03 mg/kg, according to THOMAS (1980)], it is highly toxic.

The chemical, physical, and biological properties of TCDD and its toxicology and behavior in the environment have been intensely investigated since 1970. An excellent summary of research on TCDD through 1972 is in the proceedings of an international conference held in April 1973 (*U.S. Department of Health, Education and Welfare* 1973). More recent reviews are provided by the *National Research Council of Canada* (1978), TURNER (1977), *Swedish National Research Council* (1978), and BLAIR (1973).

The extremely small quantities of TCDD which are likely to be present in environmental samples have forced the development of the most rigorous and sensitive analytical methodologies yet developed for chemicals in the environment (BAUGHMAN and MESELSON 1973, HUMMEL 1977, SHADOFF and HUMMEL 1978, O'KEEFE *et al.* 1978). These methods are now yielding data that will be of substantial value in establishing the environmental behavior of TCDD in the field and the degree to which theoretical and laboratory studies accurately predict this behavior.

a) Photochemical degradation

CROSBY *et al.* (1971) reported pure crystalline TCDD was not subject to photolysis, but PLIMMER *et al.* (1973) showed a hydrogen donor was

Table XVI. *Tissue concentrations and bioaccumulation ratios of ^{14}C-2,4,5-T and ^{14}C-TCDD in aquatic organisms as a function of time.*[a]

Organism and chemical	Initial[b] soil conc.	Time (days)						
		Adsorption[c]					Desorption[d]	
		1	3	7	15	32	39	46
Water fleas								
2,4,5-T	0.1	2.2[e] (4)[f]	4.0 (4)	1.2	21.5	27.4 (15)	—	—
	1.0	4.5 (1)	15.4 (2)	171.4	127.0	243.0 (16)	—	—
	10.0	36.9 (1)	—	3,423.9	—	1,308.7 (10)	—	—
TCDD	0.1	6.8 (2,000)	12.2 (4,207)	17.1 (7,125)	8.6 (3,308)	7.4 (1,762)	—	—
Fish[g]								
2,4,5-T —1	0.1	2.0 (3)	10.4 (12)	0.6	17.5	42.2 (23)	15.1	—
	1.0	12.3 (2)	33.1 (4)	30.0	82.5	224.8 (15)	167.2	—
	10.0	1,262.0 (10)	1,032.0 (10)	347.4	680.8	3,729.3 (26)	3,104.0	—
TCDD —1	0.1	2.3 (676)	4.8 (1,482)	11.7 (4,875)	[h]			
TCDD —2		0.9	2.3	4.1	5.0	[i]		
Algae								
2,4,5-T	0.1	3.1 (5)	3.2 (4)	10.2	52.0	87.6 (49)	57.4	20.5
	1.0	41.2 (7)	16.4 (2)	120.7	359.0	732.2 (48)	538.0	201.8
	10.0	105.5 (2)	106.4 (1)	1,072.3	5,525.1	6,125.8 (43)	5,581.1	3,769.2
TCDD	0.1	0.02 (6)	3.0 (1,034)	5.0 (2,083)	1.7 (654)	4.2 (1,000)	2.1	1.0
Snail								
2,4,5-T	0.1	1.7 (3)	3.2 (4)	2.6	10.4	10.6 (6)	8.0	5.4
	1.0	8.3 (1)	8.4 (1)	9.6	54.3	50.6 (3)	39.7	97.4
	10.0	39.8 (1)	34.0 (0)	122.5	1,641.5	585.0 (4)	805.4	836.5
TCDD	0.1	2.5 (735)	3.6 (1,241)	5.0 (2,083)	9.7 (3,731)	5.8 (1,381)	0.5	1.6

[a] From Table 2 of YOCKIM *et al.* (1978).
[b] 2,4,5-T in ppb (μg/kg), TCDD in ppt (ng/kg).
[c] Adsorption = period when organism was in ecosystem with test chemical.
[d] Desorption = period when organism was moved to untreated water.
[e] Tissue level in ppb.
[f] Bioaccumulation ratios = tissue concentration/w... concentration.
[g] Fish 1 = *Gambusia affinis* and 2 = *Ictalurus pun...us*.

Table XVII. *MCPA in a model aquatic-terrestrial ecosystem treated with MCPA at 5 kg/ha. Each value represents the mean of 2 experiments with 2 model replicates.*[a]

Substrate	Residues of MCPA (mg/kg) Days after application 3	6	12	CF[b]
Terrestrial plants	27.7	11.98	3.15	
Soil	0.196	0.096	0.114	—
Water	0.045	0.121	0.156	—
Water plants	0	0	0	0
Fish[c]	0.024	0.005	0.020	<1
Snail	0.866	0.577	0.075	<1
Mouse[d]				
Liver	0	0	0.52	3.25
Kidney	0	0	0.64	4.00

[a] From Table 2 of Virtanen *et al.* (1979).
[b] $CF = \frac{\text{concentration in the organism}}{\text{concentration in water}}$
[c] MCPA was found in only 1 fish.
[d] MCPA was found in liver and kidney of 1 mouse.

necessary, and TCDD decomposed rapidly when dissolved in methanol and exposed to UV light. Dobbs and Grant (1979) studied the photolysis of a series of chlorinated dibenzo-*p*-dioxins by sunlight and found they degrade to less toxic forms in sunlight. Gebefuegi *et al.* (1977) studied TCDD photochemical degradation under simulated environmental conditions. They reported degradation rates of 92% in longwave UV and 98% in short UV in 7 days. Their findings suggest the possibility of the photodegradation of TCDD in sufficient sunlight even in the absence of organic proton donors.

b) Vegetation—residues and fate

Crosby and Wong (1977) analyzed the persistence of TCDD in herbicide formulations on leaves, soil, or glass plates (Fig. 4). When exposed to natural sunlight (full summer sun), most of the TCDD on leaves disappeared in less than 6 hr. This loss was due principally to "photochemical dechlorination." The herbicide formulation provided a hydrogen donor which allowed rapid photolysis to occur. Norris (1980) reported rapid disappearance of TCDD from leaves even under the low light condition of a cloudy day. A half-life of 7 to 10 hr was observed when the UV energy (295 to 385 nm) input was 28 to 43 KJ/m²/hr. In contrast, full summer sunlight delivered about 150 KJ/m²/hr at this study site. The author noted, however, that volatilization as well as photodecomposition may be responsible for the disappearance noted.

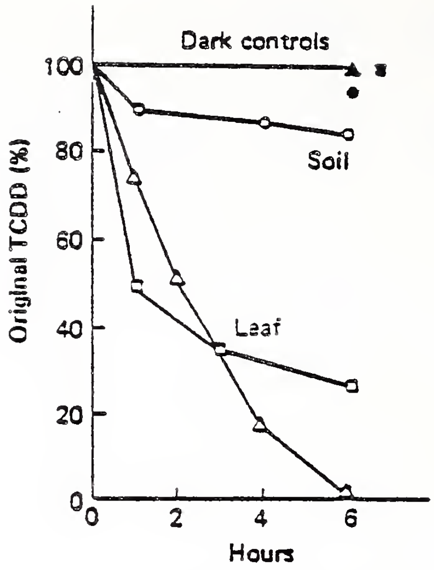

Fig. 4. Disappearance of TCDD from leaves, soil, and glass in sunlight (CROSBY and WONG 1977).

Laboratory studies indicate uptake of TCDD from soils by plants may not be significant. Soybean and oat plants took up only trace amounts of TCDD in the first 10 to 14 days after exposure to sandy soil containing 200,000 times the amount of TCDD contained in an application rate of 2.24 kg/ha of 2,4,5-T (with 0.1 mg/kg of TCDD). No detectable TCDD was in the grain or beans at maturity. COCUCCI *et al.* (1979), however, reported TCDD in leaves, twigs, fruit, and cork of several woody plants growing in soil that had been contaminated by the release of TCDD at Seveso, Italy. The twigs and cork material were present at the time of the TCDD release, but the fruit and leaves were not. The residues found in a cherry tree were: leaves—1.01 μg/kg, fruit—0.395 μg/kg, twigs—13.14 μg/kg, and cork—3.18 μg/kg. These results indicate TCDD is mobile in plants, at least to a small degree. The underground and aerial organs of carrot, potato, onion, and narcissus contained detectable levels of TCDD, but the concentration was usually less than the concentration in soil. The TCDD level in underground organs decreased markedly (10- to 30-fold decrease) in 2 to 4 mon when the plants were transplanted to unpolluted soil (Table XVIII). TCDD was not translocated from the point of application on the leaf surface to other parts of the plant, and some was washed off with rain water (ISENSEE and JONES 1971). In another field study by CHENEY *et al.* [no date], no TCDD was detected in samples of foliage collected 1, 2, 4, and 8 days after application of 3.36 kg/ha of 2,4,5-T PGBE ester. SUNDSTROM *et al.* (1979) reported only 1 of 8 samples of vegetation from 6 reforestation areas sprayed with 2,4,5-T

Table XVIII. *TCDD in soil and vegetation, Seveso, Italy.*[a]

Soil[b] ($\mu g/kg$)	Vegetation	Aerial parts	Residue in underground parts ($\mu g/kg$)			
			Inner	Outer	When transplanted to unpolluted soil	After time in unpolluted soil
5.310	Carrot	2.150	9.203	4.462	8.90	0.760[c]
8.330	Potato	2.115	1.961	3.560	1.86	No value[e]
2.680	Onion	0.835	1.763	1.798	2.26	0.062[d]
4.370	Narcissus	1.658	2.593	2.221	2.62	0.022[e]

[a] From Tables 1 and 2 of Cocucci *et al.* (1979).
[b] 10-cm diameter soil cores, 14 cm long.
[c] Transplanted without aerial parts, harvested 4 mon later.
[d] Transplanted with aerial parts, harvested 4 mon later.
[e] Transplanted without aerial parts, harvested 10 mon later.

(*n*-butyl, isobutyl ester mixture, or 2-butoxyethyl ester) contained detectable TCDD (170 ng/kg TCDD). The ratio of TCDD to 2,4,5-T was lower in the vegetation than in the formulation, which suggests more rapid disappearance of TCDD than of 2,4,5-T.

c) Soil—residues and fate

KEARNEY *et al.* (1972) indicated that pure TCDD on soil surfaces was not degraded by sunlight. CROSBY and WONG (1977) demonstrated that TCDD in formulated herbicide is rapidly degraded (about 15% in 6 hr) on the soil surface by the action of sunlight (Fig. 4). TCDD was immobile in 5 soils with widely varying properties (HELLING *et al.* 1973). The possibility of TCDD entering groundwater is remote (TSCHIRLEY 1971). If TCDD is incorporated into soil, it disappears slowly. About half the TCDD was lost after 1 yr in a laboratory study (KEARNEY *et al.* 1972). It seems unlikely, however, that TCDD would be incorporated in soils under most conditions of use, since it does not leach into the soil. TCDD is not produced from breakdown products of 2,4,5-T in soils or in sunlight (KEARNEY *et al.* 1973 a and b).

d) Water—residues and fate

TCDD is nearly insoluble in water, about 0.0002 mg/L. For this reason, it would be expected to remain on plant and soil surfaces at the application site. Because it is immobile in soils, KEARNEY *et al.* (1973 b) concluded there would be "no ground water contamination problem." These properties suggest TCDD is unlikely to enter forest streams via leaching or overland flow. Direct application or drift of 2,4,5-T or silvex to water will also result in entry of TCDD to the water. TCDD would most likely be associated with the less water-soluble constituents of a formulation which may form a thin film on water surfaces (MACKAY and WOLKOFF 1973). TCDD in such films should be rapidly degraded by sunlight, much like the thin films on vegetation and soil studied by CROSBY and WONG (1977).

WARD and MATSUMURA (1978), studying the fate of TCDD in a model aquatic ecosystem, reported a strong partition (about 95%) in favor of TCDD adsorption to sediments. TCDD half-life in the sediment (incubated in the laboratory anaerobic conditions, underwater) was about 600 days. The concentrations used in the experiment were high (0.7 to 1.3 mg/kg TCDD), more than 10^6 times greater than would be expected from normal rates of application of 2,4,5-T. The effect of concentration on rate of loss is not known. YOCKIM *et al.* (1978) also used very high levels of TCDD in sediment (0.1 mg/kg) in a model aquatic ecosystem study. They found the TCDD equilibrated between sediment and water phases in about 1 day with about 3 ng/L of TCDD in water and 0.1 mg/kg of TCDD in sediment (a 33,000:1 distribution ratio between sediment

and water). They noted an 18% decrease in TCDD level in the sediment between 30 and 150 days of incubation (if the decrease was linear with time, the half-life was 333 days in this system).

The actual levels of TCDD in vegetation, forest floor, soil, and water from the field have been measured in only a few cases. They can be estimated, however, from initial residue levels of 2,4,5-T (NORRIS *et al.* 1977) (assuming 2,4,5-T contains 0.1 ppm TCDD), and assuming the TCDD persistence characteristics reported by CROSBY and WONG (1977), KEARNEY *et al.* (1973 a and b), and MILLER *et al.* (1973) apply (Table XIX). Verification of these values is needed from actual residue studies.

e) Bioaccumulation

TCDD is present in such minute quantities in the environment that primary exposure to TCDD (that is, exposure resulting from direct ingestion, dermal absorption, or inhalation) is limited (NORRIS *et al.* 1977). Bioaccumulation is a mechanism by which organisms may collect or concentrate TCDD from primary exposure. If significant bioaccumulation occurs, these organisms (as food sources for other creatures) could possibly carry toxicologically significant residues. The question is, then, does bioaccumulation of TCDD occur and, if it does, to what degree does it occur in connection with normal patterns of use of 2,4,5-T and silvex? There are three ways to study this question: physical-chemical properties, laboratory studies, and environmental monitoring.

1. Physical-chemical properties.—Physical-chemical properties are good indicators of the potential for bioaccumulation. Chemicals with low water solubility and high fat solubility have a strong potential for bioaccumulation. DDT is an example of a chemical which is low in water solubility (0.001 mg/L), high in fat solubility (86,000 mg/L in corn oil),

Table XIX. *Calculated residues of TCDD in the forest after aerial application of 2,4,5-T (containing 0.1 mg/kg TCDD) at 2.24 kg/ha.*[a]

Time after application (days)	Vegetation (ng/kg[b])	Forest floor (ng/m²)	Soil (ng/kg[b])	Water (ng/L[b])
0	5	4	0.001	1
1	0.001[c]	0.5[c]	0.001[d]	1×10^{-4}[e]
4	—	0.004	0.0009	—
18	—	—	0.0008	—
28	—	—	0.0006	—
52	—	—	0.0005	—

[a] Calculated from NORRIS *et al.* (1977).
[b] Part per trillion.
[c] Assumes TCDD persistence reported by CROSBY and WONG (1977).
[d] Assumes TCDD persistence reported by KEARNEY *et al.* (1973 a and b).
[e] Assumes TCDD persistence reported by MILLER *et al.* (1973).

and is known to bioaccumulate in exposed organisms. TCDD is low in water solubility (0.0002 mg/L) but is also low in fat solubility (47 mg/ in corn oil). The ratio of oil solubility to water solubility is 86 × 10^6 for DDT and 0.2 × 10^6 for TCDD. These physical-chemical properties suggest that TCDD would bioaccumulate in exposed organisms but probably to a lesser degree than DDT. The degree of bioaccumulation is likely to depend on the magnitude and duration of organism exposure.

2. Laboratory studies.—Bioaccumulation can also be studied in animal feeding studies or in small laboratory ecosystems. In laboratory feeding studies involving repeated exposure to TCDD, FRIES and MARROW (1975) reported that after 6 wk of exposure, the TCDD level in rats reached a steady-state which was 10.5 times the daily intake. ROSE *et al.* (1976) also reported steady-state concentration in rats in 7 wk (slightly more then 10 times the daily intake level). In a feeding study with rainbow trout, HAWKES and NORRIS (1977) reported limited data indicating that on a whole-body basis, TCDD levels in fish were approximately of the same order of magnitude as the level of TCDD in the food which they consumed. These data indicate that animals which ingest TCDD in their diet will accumulate TCDD to body burdens about 10 times the concentration in the diet and that certain body tissues will contain residues for at least as long as exposure continues.

TCDD was not irreversibly accumulated in these feeding studies, and once exposure to TCDD stopped, the body burden decreased. PIPER *et al.* (1973 a), ALLEN *et al.* (1975), FRIES and MARROW (1975), and ROSE *et al.* (1976) all found a half-life for TCDD residence in the body which ranged from 12 to 30 days.

Several laboratory-scale aquatic ecosystem studies have been conducted with TCDD. MATSUMURA and BENEZET (1973) exposed several aquatic organisms to TCDD in such a system; unfortunately, in most instances the concentration of TCDD in the water was substantially in excess of the limits of its solubility which prevents meaningful interpretation of the data. In one experiment, however, TCDD was adsorbed on sand in the bottom of the aquariums and they found 0.0001 mg/L of TCDD in water and 0.157 mg/kg in brine shrimp, to give a concentration factor or bioaccumulation ratio of 1,570:1. The bioaccumulation ratio is the ratio of the concentration of TCDD in the organism to the concentration of TCDD in the water.

ISENSEE and JONES (1975) also used a laboratory-scale, aquatic ecosystem to study TCDD bioaccumulation in mosquito fish, fingerling channel catfish, algae, duckweed, snails, and water fleas. TCDD was adsorbed on soil which, when equilibrated with the water, resulted in TCDD concentrations in water ranging from 0.05 to 1,330 ng/L. Concentrations in excess of 200 ng/L exceed the limits of water solubility for TCDD and prevent meaningful interpretation of those bioaccumulation data. In experiments where the water concentration was less than 200

ng/L, ISENSEE and JONES (1975) reported bioaccumulation ratios ranging from 2×10^3 to 63×10^3. They found a strong positive correlation between the concentration of TCDD in tissue and concentration of TCDD in water for all organisms. The total amount of TCDD accumulated was proportional to its concentration in water, and equilibrium concentrations in tissues were reached in 7 to 15 days. They reported TCDD bioaccumulated to about the same magnitude as many of the organochlorine insecticides in model aquatic ecosystems.

YOCKIM *et al.* (1978) used basically the same technique as ISENSEE and JONES (1975). They reported generally increased TCDD accumulation with time in water fleas, fish, algae, and snails, but the levels appeared to decline when the organisms were placed in clean water for periods of 7 to 14 days (Table XVI). The bioaccumulation ratios during the period of exposure ranged from 2×10^3 to 7×10^3 with the highest ratios in water fleas and the lowest in algae. MILLER *et al.* (1979) exposed young coho salmon to TCDD in water for periods ranging from 1.5 to 96 hr and then transferred the fish to clean water. After 114 days in clean water, whole-body residues contained from 4 to 48% of the TCDD originally placed in the exposure container. The residue levels increased linearly with duration of exposure. The reason TCDD clears more slowly from aquatic organisms than terrestrial animals once exposure ceases is not known. The apparent persistence of the whole-body residues of TCDD in fish indicates analysis of fish may provide a sensitive technique for detecting the presence of TCDD in aquatic ecosystems in the field.

These results from laboratory studies indicate that organisms exposed to TCDD in their diet or in aquatic ecosystems will bioaccumulate TCDD. The degree of bioaccumulation which occurs from the use of TCDD-contaminated herbicides in natural ecosystems depends on the magnitude and duration of organism exposure. In laboratory studies, organism exposure is assured through regular addition of TCDD to the food or (in aquatic ecosystems) from a substantial reservoir of TCDD adsorbed on sand or soil which continuously releases small quantities of TCDD to water.

In the natural environment, several processes operate to reduce or eliminate TCDD exposure to organisms and thereby minimize the opportunities for bioaccumulation. CROSBY and WONG (1977) reported TCDD in herbicide formulations disappeared rapidly from vegetation and soil when exposed to sunlight. This mechanism would markedly reduce or eliminate organism exposure through dermal contact with or ingestion of contaminated vegetation. In the aquatic environment, the likelihood of 2,4,5-T and TCDD entry to aquatic systems is slight; but if it does occur, chemicals in the water are rapidly diluted and carried downstream with streamflow. TCDD which adsorbs on sediments provides a reservoir of TCDD in the aquatic environment similar to that provided in the model aquatic ecosystem studies. In real stream systems, TCDD desorbed from

sediments would be quickly moved downstream with streamflow, thus the opportunity for bioaccumulation by a particular organism would be limited.

3. **Environmental monitoring.**—The third approach to evaluating TCDD bioaccumulation is to look directly for evidence of bioaccumulation in the field. Several efforts have been made but with markedly different levels of sophistication and sensitivity of analytical methods. For instance, WOOLSON *et al.* (1973) analyzed samples of eagle tissues from various regions in the United States. No TCDD was detected, but the minimum detection limit of 50 μg/kg is not adequate to properly evaluate bioaccumulation of TCDD. Similarly, GARCIA and RHODES (1979) did not detect TCDD in American coots collected near a lake which received runoff from rangeland areas sprayed with 2,4,5-T. They did not specify their minimum level of TCDD detection, but it appeared to be about 1 μg/kg.

YOUNG *et al.* (1976 and 1978) studied the behavior and bioaccumulation of TCDD in animals from the Eglin Air Force Base site used for equipment development and testing for application of herbicides in Vietnam. The study area received massive applications (500 to 1,000 kg/ha) of 2,4,5-T, much of which contained TCDD in excess of 1 mg/kg. Analysis of soil from the test site showed TCDD residue levels in the range of 10 to 1,500 ng/kg several yr after residue applications had stopped. Analysis of rodents, reptiles, birds, fish, and insects shows the presence of TCDD in tissues of at least some of the organisms from this area. The level of TCDD in the herbicide used at the Elgin test site and the massive rates of application make these data not directly applicable to the registered uses of 2,4,5-T and silvex in the United States.

Studies done in connection with the registered uses of 2,4,5-T for vegetation control have found relatively little TCDD in biological samples. In 1973–74, the Environmental Protection Agency, cooperatively with the USDA Forest Service, conducted a monitoring program for TCDD in tissues of animals from several areas which had been recently treated with 2,4,5-T in western Oregon and Washington. The analytical methodology employed, however, was not adequate to establish the presence of TCDD in those environmental samples. It was adequate to determine which samples did not contain TCDD in the low-to-mid part-per-trillion range. Approximately 84% of the samples did not contain detectable levels of TCDD. The remaining samples were described by EPA as "minutely suggestive" for TCDD. In 1976, 5 of these "possible positive" samples were reanalyzed by 2 laboratories (participants in the dioxin monitoring program); 2 samples did not contain detectable TCDD. EPA described the results of analysis of the other 3 as follows: "Some of the samples analyzed in 1973–74 still appear positive for TCDD. Unfortunately, the results from the two laboratories participating in the confirmation vary widely. The confirmation analysis, therefore, still does not give a precise quantification of the amount of TCDD

present. It does appear, however, that from a qualitative standpoint TCDD was present in a small percentage of the forest samples collected in 1973." Assuming 3 out of 5 samples (60%) which were possible positives in the 1973–74 analysis are, in fact, qualitative for TCDD, then 9.6% of the 1973 samples were positive for TCDD and 90.4% did not contain detectable residues (*U.S. Department of Agriculture, Forest Service* 1978 c).

NORRIS (1980) reported first results from a survey of 6 Oregon forest stream systems for TCDD. More than 80 samples of fish from 23 sampling stations were analyzed to a detection limit of less than 10 ng/kg. All the fish were negative for TCDD at 10 ng/kg, but 5 (one a control) were reported as possible positives at 0.2 to 0.4 ng/kg greater than the limits of detection. Of the 6 stream systems surveyed, 4 have an extensive history of 2,4,5-T application.

The U.S. Environmental Protection Agency (EPA) beef-fat-monitoring program which was initiated in 1974 has been completed. Samples of beef fat (85) and liver (43) have been analyzed for TCDD. Approximately 25% of these samples are from animals not exposed to areas sprayed with 2,4,5-T. The *U.S. Environmental Protection Agency* (1977 a) reported that 1 sample showed a positive TCDD level at 60 ng/kg and 2 at 20 ng/kg; 5 samples appeared to have TCDD in the range of 5 to 10 ng/kg. EPA stated, "The analytical method is not valid below 10 ppt, although a recent dioxin implementation plan meeting statement set 9 ppt as the minimum detectable level." Of the 43 liver samples analyzed, 1 sample might contain TCDD, but the level is too close to the sample detection limits for quantification. A fat sample from the same animal showed no TCDD residue. The results of the EPA beef-fat-monitoring study indicate bioaccumulation of TCDD in grazing animals is not sufficient to result in regularly detectable levels of TCDD (>10 ng/kg) in beef fat and liver.

NEWTON and SNYDER (1978) reported on the analysis of livers from mountain beavers (*Aplodontia rufa*) captured 2 mon after a forested area in western Oregon was treated with 2,4-D and 2,4,5-T. Mountain beavers have a small range and are herbivorous, thereby affording them substantial exposure to herbicide-treated plants. In addition, they are a burrowing animal which will put them in intimate contact with herbicide and TCDD present on the soil surface. Analysis showed no detectable levels (<10 ng/kg) of TCDD. SHADOFF *et al.* (1977) looked for TCDD accumulation in animals due to the use of 2,4,5-T and silvex in the midwestern United States. They did not detect any TCDD (detection limit about 10 ng/kg) in samples of fish, water, mud, and human milk from areas in Arkansas and Texas.

MESELSON and O'KEEFE (1977), in a preliminary report to Oregon Congressman Weaver, indicated some samples of human milk from residents of areas in which 2,4,5-T was used contained detectable levels of TCDD (3 samples out of 6 from Texas, and 1 sample out of 5 from

Oregon). The levels detected were at the limits of detection and were substantially below the 10 ng/kg level established by EPA in the beef-fat-monitoring program as the minimum acceptable, reportable level. Recently, the *U.S. Environmental Protection Agency* (1980) announced results of analysis of samples of milk from 105 human beings selected from areas in California, Oregon, and Washington where the dioxin-containing herbicides were known to have been used for several yr. No residues of TCDD were detected at levels of detection that varied from 1 to 4 ng/kg.

4. Conclusions about bioaccumulation.—The results of these various tests indicate that if TCDD is present in the environment in a form which is available to organisms, then bioaccumulation will occur if organisms are exposed. The degree to which bioaccumulation of TCDD occurs in the field is dependent not only on the physical-chemical properties of the compound but also on the persistence and availability of TCDD in the environment.

Monitoring efforts indicate that substantial bioaccumulation of TCDD (sufficient to produce residue levels in excess of 10 ng/kg of TCDD in the majority of the population) is not occurring in animals in or near areas treated with 2,4,5-T or silvex in operational programs. Apparently mechanisms of degradation and dilution which operate in the natural environment reduce the opportunities for organisms to be exposed, and thereby reduce the degree to which TCDD bioaccumulation occurs.

This conclusion is not in conflict with recently reported findings of TCDD in fish from the Titawabasee River downstream from the Dow Chemical Co. manufacturing plant at Midland, Michigan (*Dow Chemical Co.* 1978 a). The residues in the fish, whether they are from plant discharge water or are from the products of combustion (*Dow Chemical Co.* 1978 b), did not result from the use of 2,4,5-T as an herbicide.

f) Thermal conversion of 2,4,5-T to TCDD

It is possible to produce TCDD on heating or burning of 2,4,5-T or 2,4,5-T-treated materials in laboratory tests. The conditions of combustion and herbicide concentration are crucial. The tests reported by BUU-HOI *et al.* (1971), LANGER *et al.* (1973 a), and others showed TCDD formation occurred when 2,4,5-T was heated in a closed container under alkaline conditions such that the sodium salt of trichlorophenol was a significant degradation product. The amount of herbicide employed in these tests was very high. LANGER *et al.* (1973 b) indicated control of the decomposition reaction to produce trichlorophenol was necessary since heating above the decomposition point (300°C) produced no TCDD. AHLING *et al.* (1977) studied both the destruction of 2,4,5-T and the formation of polychlorinated dibenzo-*p*-dioxins. They noted 99.995% decomposition of the 2-butoxyethyl ester of 2,4,5-T maintained at 400° to 500°C for 2 sec. However, about 3 mg of one or more 4-chlorodibenzodioxin isomers were

formed for each kg of formulation burned. The emission decreased with increased temperature and time. At 850°C and at times greater than 0.9 sec, none, or barely detectable levels of 4-chlorodibenzodioxin were formed.

Concentration of herbicide is important because the formation of TCDD from chlorophenols is apparently a bimolecular reaction. If conditions of heat and alkalinity are conducive to the condensation of the phenol to form TCDD, then the extent of condensation varies with the number of molecules available to interact with one another. In the experiments reported by AHLING *et al.* (1977), the concentration of 2,4,5-T (10,000 mg/kg of wood chips) was about 50 to 500 times higher than levels reported in leaves.

Experiments such as those of LANGER, AHLING, and others are useful because they show that thermal production of TCDD is chemically possible. They drastically overestimate, however, the levels of TCDD which might be produced by burning in the field because in the laboratory tests (1) the concentrations of herbicide were several times greater than the levels of 2,4,5-T which occur in the field and (2) heating was prolonged, uniform, and combustion did not always occur. Temperatures at which thermal decomposition of TCDD occurs (800°C) were not usually attained in these tests. Actual burning, of course, will produce temperatures as low as those used in laboratory tests only briefly. As burning temperatures approach 800°C, thermal decomposition of TCDD occurs (AHLING *et al.* 1977). When combustion can take place with a free exchange of air, temperatures above 1,200°C are common. Under these conditions, complete oxidation of 2,4,5-T, trichlorphenols, TCDD, and similar chemicals is expected to occur.

There are only limited experimental data on how much TCDD is produced when 2,4,5-T is burned. WATTS and STORHERR (1973) noted burning and heating of such 2,4,5-T-treated products as vegetation, meat, and fat did not produce detectable TCDD. The sensitivity of their analysis, however, was not adequate to detect environmentally important quantities of TCDD. The most pertinent data came from a laboratory experiment in which grass treated with 2,4,5-T at 12 lb/A was burned under conditions somewhat resembling those which might occur in the field (STEHL and LAMPARSKI 1977). Their study showed an approximate 0.00016% conversion of 2,4,5-T to TCDD. This involved a semiclosed system, however. Thus, any TCDD which might normally have been lost to the air as vapor or adsorbed on smoke particles in forest burning was captured and retained in this system.

The amount of TCDD likely to be produced on burning treated vegetation is dependent on the concentration of 2,4,5-T in the vegetation. NORRIS *et al.* (1977) determined the persistence of 2,4,5-T in Oregon forests. Calculated levels of TCDD which might be produced by burning vegetation containing 2,4,5-T according to NORRIS *et al.* (1977) and assuming the conversion ratio reported by STEHL and LAMPARSKI (1977)

Table XX. *2,4,5-T residues on vegetation (measured) and TCDD (calculated) that might be produced by burning vegetation.*

Mon after application	2,4,5-T[a] (mg/kg)	Possible TCDD level if burning occurs at time indicated[b] (ng/kg)
0	95	152
1	9.1	14
3	0.10	0.16
6	0.07	0.11
12	0.01	0.02

[a] From NORRIS *et al.* (1977).
[b] Assumes conversion is 0.00016% (STEHL and LAMPARSKI 1977).

are in Table XX. AHLING *et al.* (1977) calculated the formation of 4 chlorodioxins (presumably including TCDD) could be as high as 1 $\mu g/m^2$ in a forest fire directly after application of herbicide. They believe, however, the amount is likely to be smaller because the concentration of 2,4,5-T would normally be less. The data of NORRIS *et al.* (1977) and STEHL and LAMPARSKI (1977) indicate levels of 0.02 $\mu g/m^2$ if burning occurs immediately after application and <0.002 $\mu g/m^2$ if burning occurs 1 mon later.

RAPPE (1978) (reporting the unpublished research of Anderson *et al.*) noted that burning "sprayed and spiked" vegetation at 600°C did not produce TCDD (<4 mg TCDD/kg of 2,4,5-T) in combustion gases, soot particles, and ash. The degree of conversion was <1 mg of TCDD/kg 2,4,5-T at 550° to 950°C in another study (unpublished research of AHLING *et al.*) according to RAPPE (1978).

Clearly, the amount of TCDD produced by combustion of sprayed vegetation depends to a major degree on when burning occurs after treatment. 2,4,5-T is occasionally used to desiccate brushfields prior to burning. Burning which takes place from 1 to 3 mon after the application could result in TCDD levels of 14 and 0.2 ng/kg, respectively. In some brush types, burning is delayed for 12 mon or more. Immediately after application the level of TCDD present on the vegetation may be about 10 ng/kg, assuming the 2,4,5-T contained 0.1 mg/kg TCDD. Data of CROSBY and WONG (1977) indicate the TCDD originally applied will be mostly gone within 1 mon of the application. Therefore, the levels of TCDD which might be produced are not expected to substantially exceed TCDD levels present as a result of the original application of herbicide.

Preliminary results from the *Dow Chemical Co.* (1978 a and b) indicated several dioxin isomers may be formed in trace amounts during the combustion of many substances (not contaminated with or associated with

2,4,5-T). Fossil fuels, automotive exhaust, trash burners, cigarette smoke, and charcoal-grilled meats have all been found to produce or contain minute quantities of various dioxin isomers, including in some cases the 2,3,7,8-tetrachloro isomer. These sources of dioxins are not associated with the registered uses of 2,4,5-T as an herbicide in any way.

Summary and conclusions

The phenoxy herbicides have the most extensive scope and longest history of use of any group of pesticides in forestry. There is extensive literature on the behavior of the phenoxy herbicides, but their use in forestry remains controversial. Risk assessment is the predominant issue.

Despite the large number of citations in this review, not all aspects of this topic have been fully cited. The greatest strengths in the literature involve 2,4,5-T movement and persistence in the major forested regions of the United States. Both the magnitude and duration of exposure that organisms are likely to receive (via vegetation, forest floor, soil, or water) can be estimated with the available data. Despite the fact 2,4-D is used far more extensively than 2,4,5-T, it has received substantially less research attention. Public and regulatory pressure to ban this chemical is increasing and, as there has been with 2,4,5-T, there is likely to be an upsurge in research activity. Dichlorprop, silvex, MCPA, and other phenoxy herbicides have received little attention in forest environment research; but their scope of use is limited and the data for their di- and tri- chlorinated phenoxyacetic acid relatives provide a reasonable basis for prediction.

In forest applications, phenoxy herbicides are distributed initially among 4 major compartments: air, vegetation, forest floor and soil, and water. Most of the herbicide ends up in the forest floor and soil. In all compartments, the phenoxy herbicides are relatively rapidly decomposed, and there appears to be little opportunity for substantial exposure to these chemicals. A fairly extensive data base is available to establish the specific levels of phenoxy herbicide likely to occur in any one location at a particular time after application.

TCDD, the highly toxic contaminant of 2,4,5-T and silvex, has also been intensively studied, but little of this work has been done in the forest. The initial levels and persistence of TCDD in specific forest compartments can be estimated from laboratory and field studies. Bioaccumulation of TCDD is expected to occur if organisms are exposed to a substantive, continuing source of TCDD, but it has not been detected (extensively) from normal patterns of herbicide use.

The most glaring deficiencies in the data base for the phenoxy herbicides involve (1) residues and fates of herbicides in air, (2) fates of herbicides in forest streams (including sediment and biota), and (3) residues and fates of TCDD in all forest compartments. Attention needs to be directed toward filling these gaps to insure that a complete data

base is available for assessing the risks which may be associated with the use of these chemical tools by forest land managers.

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Questions Proposed by Managers

SOIL

The state of the art in scheduling of timber harvest to minimize cumulative effects (at least in NFS) is to restrict harvest to no more than a certain percentage of the gross watershed area for a given period of time. There are, no doubt, better variables to use when scheduling timber harvest than simply gross harvest area. For example, the miles of mid-slope road is surely a better variable than total road miles. Acres clearcut on steep or unstable terrain is surely better than total acres clearcut.

1) What variables should be used or management activities constrained while scheduling timber harvest?

2) What variables are the best indicators that cumulative effects are occuring? An increased frequency of landslides? Changes in stream channel morphology, like aggradation or degradation? Increased peak discharges? Reduction in the Fredle Index for anadromous spawning gravels? Increase in basin stream water temperature?

3) What are the relative impacts upon the soil of live skyline yarding to a landing compared to lateral yarding?  

4) Can heavy fuels concentrations be broadcast burned during brushfield conversions without unacceptable soil-water damage? What is the best measure of site deterioration?

5) Discuss the relative contribution to annual sediment yield from mass movement compared to surface and channel erosion for southwestern Oregon.

6) What is the contribution to site productivity from soil duff layer and branch/needle slash in both high and low elevation western Cascades?

7) What is the future of watershed research on the South Umpqua Experimental Forest?

8) It appears, from our observations of various roaded watersheds, that most of the sediment entering these systems is road related. The culprits are roads close to stream courses with poor drainage. It appears that of these, the main "polluters" are insloped roads in erosive soils with long distances between culverts.

a) Since the ideal road design for water quality and slope stability is that which does not significantly change the natural movement of water on a slope, is not a properly maintained outslope road the "best" road design?

b) If so, what is the maximum road gradient that outsloping should be designed for? Why? What slope gradients? What precautions or maintenance measures are needed?

c) If inslope roads is your answer for steeper road gradients, culvert spacings will need to be closer. This of course is very expensive and usually the first thing reduced when road money is cut. With this in mind how do inslope roads compare to outslope roads? It would be nice to see a comparison chart showing impacts and benefits of each.

d) Insloped roads pick up sediment that has fallen into the ditch and transports it to stream system. What, if any, are the best stabilizing measures (i.e., structures, vegetation - what kinds? are they effective?).

9) There has been a lot of discussion on the importance of nitrogen on high elevation soils and soils on hot, dry sites. How important is duff and big fuels (1000 hrs.) to nitrogen maintenance and site productivity? To obtain a "cool" burn while meeting reforestation and fire objectives sometimes means reducing the YUM spec. Can this be detrimental to the site? Harvey et al has shown the importance of big fuels on harsh sites for nitrogen fixation in the Rockies. Does this hold true in the Siskiyou's - high elevation and south slopes? If so, should we be leaving more big fuels on the ground?

10) Has research done any additional work on lands classified as U3 - Unsuitable due to Instability or Instability Potential? Specifically, clarification of this Land Management Planning designation is

needed. Forest specialists (engineering geologists, soil scientists, hydrologists) need "back up" from research regarding the necessity for including the "influence area" of a U3 landslide in the U3 classification.

11) The economics of road building (and future maintenance and reconstruction) and logging in landslide terrain is a subject requiring research support at this time.

12) Could research make available, or inform Forest where they are available, the following:

a) Bibliography - by Author - of publications pertinent to Forest Road Construction and Logging Impacts.

b) Bibliography - by Subject - of publications pertinent to Forest Road Construction and Logging Impacts.

WATER

1) Could you provide a comparison on the persistences and mobility in the environment of 2,4-D, Triclorpyr, Picloram, Round-up Atrozine Dalapon, and the dyes rhodamine and fluorescein?

2) This is more of a philosophical question but under what circumstances is a water monitoring program for herbicides indicated?

3) What are the acceptable levels (standards) for herbicides in forest streams and responsibilities for assuming levels in the absence of standards?

4) Harr's recent paper on timber harvest decreasing water yield in the Bull Run watershed shakes our longstanding belief about water yield and removal of forest vegetation? What is the areal extent of the fog zone in western Oregon and Washington, and the implications for timber harvest scheduling?

5) Clearcutting in the transient snow in theory is capable of increasing peak discharges by more rapid melting of the deeper snow in openings. In practice, however, research has not documented the phenomenon for larger peaks at least in small barometer watersheds. Could you comment on this? What are the areas and elevations where the transient snow zone exists?

FISHERIES

The value of large organic debris in anadromous fish-bearing streams is becoming known. The large debris apparently plays a role in the formation of pools and provides a refuge for the fish. But what is the role of woody debris both large and small in first, second order or other non-fish bearing streams? In steep channels where do the benefits of nutrients contributed by logging debris outweigh the risk of debris torrents?

1) It is very difficult for the NFS, given our limited manpower and budgets, to separate natural and man-caused effects when measuring suspended sediment. Moreover, we often assume that more sediment has lasting adverse effects on anadromous fish. Instead of monitoring suspended sediment, why not monitor the quality of fish habitat over time to determine the effect of management activity. After all, this is the beneficial use we are most often concerned with. For example, could a 3 factor indicator of habitat quality be developed that would integrate the most often limiting habitat requirements of fish - quality and quantity of rearing pools, water temperature, and quality of spawning gravels?

2) On the Siskiyou Forest rearing populations of juvenile coho salmon exist only in those streams having dense canopy cover and related cooler summer water temperature. Literature indicates that juvenile coho rearing habitat requirements, at least temperature, are similar to steelhead. Our field experience in S.W. Oregon has shown coho to be much less tolerant of warm water than steelhead.

3) A lot of emphasis has been placed on total sediment yield. From the fisheries standpoint the survival of eggs and fry is highly dependent upon time of turbidity and sediment occurrence. Pioneer road construction, during winter months, has resulted in fish habitat problems, yet this type of activity continues over much of the forest.

4) To what extent has the introduction of hatchery stock reduced disease resistance of anadromous fish populations?

5) What is the probability of heavy annual stream sedimentation having major influence on water temperature of S.W. Oregon streams? Several biologists have expressed their belief that summer water temperatures are showing influence on increased intra-gravel sedimentation which has forced more water to the surface and exposure to direct solar radiation.

6) There has been an effort to reduce purchaser credits by not putting rock on spur roads. Erosion is likely to be greater on these unprotected roads. What is the potential impact of that erosion on downstream fish habitat, and what is the cost of that impact? It is really cost-effective to not rock spur roads from a fish habitat perspective?

7) How will reduced road maintenance efforts on most Forest roads affect their erosion characteristics; what will be the impact on fish habitat? Is the money saved worth the effect on fisheries?

Mike Bauer

FISHERIES

Literature contains good information on the problems that sediment causes during incubation. Generally the effects are measured in terms of dissolved O_2 concentrations. Reduced D. O. levels caused by sediments in the gravel can reduce emergence of fry. Since this is perhaps the most important problem caused by sediment (reduced D.O.) it would seem like a desirable parameter to monitor. Bob Phillips, once told me that measuring D. O. in the subsurface flows is difficult.

Since I am grasping for information on the condition of spawning gravels, I would like a procedure that could be fairly easily used by District personnel to assess this habitat. Reiser and Bjornn mention studies showing that substrates containing 20-25%+ of sediments less than 6.4 ~~mar~~ mm caused reduced emergence in steelhead and salmon. A number of questions regarding this parameter are as follows:

-Could the percentage of fine sediments be used to roughly calculate a reduction in emergence?

-Could the percentage of fine materials in the gravel be easily measured by District personnel without expensive equipment?

-Are the percentages measured in weight or volume?

-Are there other better methods available for measuring "health" of gravels?

With ~~estensive~~ extensive roading we are seeing increased sediment loads in the small streams. We talk about the impacts of sedimentation in a subjective manner, but we really need a quantitative, objective method for assessing the sedimentation problems.

In regard to roading, are there any studies of the quantitative effects of roading on fish populations? If so perhaps you could elaborate on the quantitative effects found, the types of roads, their location relative to the streams, road drainage methods etc.

Managing timber adjacent to streams always seems to generate conflict between foresters, biologists, hydrologist, etc. Some would prefer no management of the timber; some would like to cut all conifers then regenerate deciduous species. Perhaps some would like to clearcut in small blocks. Do you believe a selective harvest would be best for most streams? This would provide some opening of the canopy to stimlate regeneration of perhaps conifers and/or deciduous. Streambank stability and shade would be considered. Will this approach, done with proper constraints lead to greater stability in the stream and its channel as opposed to treatment of a greater magnitude or no treatment?